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WEST VIRGINIA ORDNANCE WORKS: ENDANGERMENT ASSESSMENT FOR
ACIDS AREA/YELLOW WATER RESERVOIR, RED WATER RESERVOIRS,
AND THE POND-13/WET WELL AREA
DRAFT FINAL REPORT

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Prepared for:

U.S. ARMY TOXIC AND HAZARDOUS MATERIALS AGENCY
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Aberdeen Proving Ground, MD 21010-5401

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<p>This report assesses endangerment to public health, welfare, and the environment posed by hazardous materials found in specific areas associated with the former West Virginia Ordnance Works (WVOW) near Point Pleasant, WV. The land, now owned by the state of West Virginia, currently comprises the Clifton F. McClintic State Wildlife Station (McClintic). The land is managed to support migratory game birds and uplands game, and public hunting and fishing are encouraged. As a result of previous investigations by the state and the U.S. Environmental Protection Agency (EPA), the site was placed on the National Priorities List under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). The areas addressed in the study, referred to as the second operable unit, included the Acids Area/Yellow Water Reservoir, the Red Water Reservoirs, and the Pond 13/Wet Well Area.</p> <p>The report summarizes and interprets findings of the remedial investigation regarding</p>			
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Block 18 (Continued)

nitroaromatics

CERCLA

2,4-DNT

2,6-DNT

2,4,6-DNT

Acids Area/Yellow Water Reservoir

Red Water Reservoirs

Pond 13/Wet Well Area

Block 19 (Continued)

source area characterization; the environmental setting; contaminant chemical and toxicological properties; and contaminant distribution in soils, sediments, ground water, and surface water.

The potential for, and rates of, contaminant migration are evaluated. The characteristics of the exposed population of wildlife and humans are addressed. Safe levels of residual contamination are determined to provide guidance for the feasibility study that will evaluate remedial alternatives. Finally, the risks posed by the hazardous materials are characterized.

The principal contaminants found sitewide at WVOW are nitroaromatic contaminants associated with TNT production. At the Acids Area/Yellow Water Reservoir, minor lead contamination is present in the surficial soils. Significant exposure pathways include direct contact with surficial soils at the Acids Area/Yellow Water Reservoir, ground water/drinking water at the Red Water Reservoirs, and ground water discharge to surface water to aquatic biota at the Pond 13/Wet Well Area.

The principle conclusions of the assessment are that no human populations are currently exposed to unsafe levels of contamination. In the future, however, if land or water use patterns changed, unacceptable exposures could result if no remedial actions were taken.

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LIST OF ACRONYMS AND ABBREVIATIONS

AAAP	Alabama Army Ammunition Plant
ACGIH	American Council of Government Industrial Hygienists
ACL(s)	alternate concentration limit(s)
ADI(s)	acceptable daily intake(s)
AIC(s)	acceptable intake(s), chronic
ASTM	American Society for Testing and Materials
BCF(s)	bioconcentration factor(s)
CaCO ₃	calcium carbonate
CEC(s)	cation exchange capacity (capacities)
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cfs	cubic foot/feet per second
CIIT	Chemical Industry Institute of Toxicology
cm ²	square centimeter(s)
cm/sec	centimeter(s) per second
DNA	deoxyribonucleic acid
DNB(s)	dinitrobenzene(s)
1,3-DNB	1,3-dinitrobenzene
DNR	Department of Natural Resources
DNT(s)	dinitrotoluene(s)
2,3-DNT	2,3-dinitrotoluene
2,4-DNT	2,4-dinitrotoluene
2,5-DNT	2,5-dinitrotoluene
2,6-DNT	2,6-dinitrotoluene

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3,4-DNT	3,4-dinitrotoluene
3,5-DNT	3,5-dinitrotoluene
EA	Endangerment Assessment
EPA	Environmental Protection Agency
EPRI	Electrical Power Research Institute
ESE	Environmental Science and Engineering, Inc.
FS	feasibility study
ft	foot/feet
ft ²	square foot/feet
ft/day	foot/feet per day
ft/yr	foot/feet per year
ft ³ /yr	cubic foot/feet per year
ft-MSL	foot/feet above mean sea level
gal	gallon(s)
gal/hr	gallon(s) per hour
gpm	gallon(s) per minute
GC/EC	gas chromatography/electron capture
GI	gastrointestinal
g/day	gram(s) per day
gpm	gallon(s) per minute
HI	hazard index
in/yr	inch(es) per year
kg	kilogram(s)
kg/day	kilogram(s) per day
kg/L	kilogram(s) per liter



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kg/m ³	kilogram(s) per cubic meter
L	liter(s)
lb	pound(s)
L/day	liter(s) per day
L/kg	liter(s) per kilogram
m	meter(s)
m ²	square meter(s)
m ³	cubic meter(s)
McClintic Wildlife Station	Clifton F. McClintic State Wildlife Station
m/day	meter(s) per day
m ³ /day	cubic meter(s) per day
m/yr	meter(s) per year
m ³ /yr	cubic meter(s) per year
meq/100 g	milliequivalent(s) per 100 grams
MG	million gallon(s)
mg	milligram(s)
mg/kg	milligram(s) per kilogram
mg/kg/day	milligram(s) per kilogram per day
mg/L	milligram(s) per liter
mg/m ³	milligram(s) per cubic meter
MG/ft ² /yr	million gallon(s) per square foot per year
MG/yr	million gallon(s) per year
mi	mile(s)
mL/g	milliliter(s) per gram
MOU	Memorandum of Understanding

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mph	mile(s) per hour
NCI	National Cancer Institute
NGVD	National Geodetic Vertical Datum
NIOSH	National Institute for Occupational Safety and Health
NIPDWR	National Interim Primary Drinking Water Regulations
NOAEL(s)	no-observed-adverse-effect level(s)
NPL	National Priorities List
NTIS	National Technical Information Service
NTP	National Toxicology Program
O.C.E.	Office of the Chief of Engineers
OSHA	Occupational Safety and Health Administration
PAH(s)	polynuclear aromatic hydrocarbon(s)
PPLV(s)	preliminary pollutant limit value(s)
ppm	part(s) per million
QA	quality assurance
QA/QC	quality assurance/quality control
RCRA	Resource Conservation and Recovery Act
RI	remedial investigation
RI/FS	remedial investigation/feasibility study
RMCL	recommended maximum contaminant level
RNA	ribonucleic acid
SPFFLV(s)	single-pathway preliminary pollutant limit value(s)
SR	State Road
SRI	Stanford Research Institute
tDNT	technical-grade dinitrotoluene

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TLM(s)	median tolerance limit(s)
TLV(s)	threshold limit value(s)
TNB(s)	trinitrobenzene(s)
1,3,5-TNB	1,3,5-trinitrobenzene
TNT	trinitrotoluene(s)
2,4,6-TNT	2,4,6-trinitrotoluene
torr/M	torr per molar
TSP	total suspended particulate
µg/day	microgram(s) per day
µg/dL	microgram(s) per deciliter
µg/g	microgram(s) per gram
µg/kg	microgram(s) per kilogram
µg/kg/day	microgram(s) per kilogram per day
µg/L	microgram(s) per liter
µg/m ³	microgram(s) per cubic meter
µg/mL	microgram(s) per milliliter
µg/100 mL	microgram(s) per 100 milliliters
USAMBRDL	U.S. Army Medical Bioengineering Research and Development Laboratory
USAMRDC	U.S. Army Medical Research and Development Command
USATHAMA	U.S. Army Toxic and Hazardous Materials Agency
USGS	U.S. Geological Survey
USSCS	U.S. Soil Conservation Service
WVOW	West Virginia Ordnance Works
WWII	World War II

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EXECUTIVE SUMMARY

Environmental Science and Engineering, Inc. (ESE), under contract with the U.S. Army Toxic and Hazardous Materials Agency (USATHAMA), is conducting a remedial investigation/feasibility study (RI/FS) of the former West Virginia Ordnance Works (WVOW). The purpose of the RI/FS is to assess soils, surface water, and ground water contamination resulting from past trinitrotoluene (TNT) manufacturing operations and to evaluate potential remedial action alternatives.

This endangerment assessment (EA) report is a component of the feasibility study (FS). The purpose of the EA is to summarize and interpret remedial investigation (RI) data, identify contaminant transport pathways and receptors, and assess actual or potential harm to public health, welfare, or the environment from hazardous materials originating on WVOW. The EA provides a transition from the RI (ESE, 1986b) to the FS which is in progress. In essence, it justifies the need for remedial action and serves to focus remedial action alternatives.



The WVOW site is approximately 6 miles north of Point Pleasant, WV, on the east bank of the Ohio River. TNT was produced at WVOW from 1942 to 1945. After World War II (WWII), the plant was closed and surface structures were razed. The industrial portion of the site was deeded to the State of West Virginia, with the stipulation that the site be used for wildlife management. If the land were to be used for any other purpose, or in the event of national emergency, the ownership of the land would revert to the Federal Government. The land, now owned by the state, currently comprises the Clifton F. McClintic State Wildlife Station (McClintic Wildlife Station) and is managed by the State of West Virginia Department of Natural Resources (DNR). DNR's management

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practices are primarily designed to promote wetlands habitats and populations of resident and migratory waterfowl. Consistent with this objective, more than 30 shallow ponds have been constructed since cessation of military activities on the site, and most of the ponds are stocked with bass. The area is open for public hunting and fishing.

Surface water contamination was observed on the site (at Pond 13) in May 1981. Following preliminary investigation by DNR and U.S. Environmental Protection Agency (EPA), WVOW was listed on the National Priorities List (NPL) of uncontrolled hazardous waste sites under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980. Under a Memorandum of Understanding (MOU) between the Department of Defense (DOD) and EPA, USATHAMA was designated the lead agency for the CERCLA response and this report is one element of the CERCLA response.

As documented in the RI report (ESE, 1986b), nitroaromatic contamination of soils, sediments, and surface waters was found in several areas of the site. 2,4,6-trinitrotoluene (2,4,6-TNT) and 1,3,5-trinitrobenzene (1,3,5-TNB) are the dominant nitroaromatic constituents, whereas the dinitrotoluenes comprise not more than 2 to 5 percent of the total nitroaromatic contamination. The dinitrotoluenes are probable carcinogens and are believed to be more toxic than either TNT or trinitrobenzene (TNB). Data collected during the original RI were determined to be sufficient to assess endangerment and conduct the FS in three areas (TNT Manufacturing Area, Burning Grounds, and industrial sewerlines), which are designated the first operable unit. The endangerment associated with the first operable unit was assessed and reported by ESE (1986b). In other areas (Red Water Reservoirs, Acids Area/Yellow Water Reservoirs, and Pond 13), denoted the second operable unit, the data developed during the original RI were inadequate to define either the extent of contamination or the potential for migration to potential receptor populations. In these areas a second phase of remedial investigations was conducted to fill identified data gaps (ESE, 1987b).

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This report assesses endangerment to public health, welfare, or the environment associated with the second operable unit.

In addition to nitroaromatic contamination, which is the characteristic type of contamination observed at the site, contamination of soils by lead was observed in the Acids Area/Yellow Water Reservoir Area.

A quantitative dose-response relationship for lead is not well established. Young children are the population at greatest risk with respect to lead exposure, and neither the current or probable future land use results in significant exposure of children to soil contaminants in this area. The proposed recommended maximum contaminant level (RMCL) for lead (EPA, 1985b) implies an allowable exposure by the drinking water pathway of 20 micrograms per day ($\mu\text{g/day}$) for young children who are assumed to consume 1 liter (L) of water per day. Based on research results by Binder *et al.* (n.d.), incidental soil ingestion by children is approximately 250 milligrams per day (mg/day). At the maximum observed soil concentration of 100 milligrams per kilogram (mg/kg), lead exposure would be 25 $\mu\text{g/day}$, approximately equal to the allowable exposure if the area were an active play area. The area is not used as a play area under the existing or probable future land use, so the observed levels of contamination do not pose an unacceptable risk.

The environmental fate properties of nitroaromatic contaminants are presented in Sec. 2.1 of this report. Key findings are that adsorption to soils and aquifer materials tends to retard migration. In subsurface systems, the nitroaromatics are relatively persistent and do not degrade very rapidly. In shallow surface water systems, however, photolysis tends to transform 2,4,6-TNT into more volatile compounds, including 1,3,5-TNB, and these compounds are volatilized to the atmosphere in a few days or weeks. There is no evidence that 2,4,6-TNT transforms to dinitrotoluenes (DNTs) in the environment.

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The toxicity of site contaminants is reviewed in Sec. 3.0 of this report. The dinitrotoluenes [2,4-dinitrotoluene (2,4-DNT) and 2,6-dinitrotoluene (2,6-DNT)] are probable carcinogens whose carcinogenicity has been confirmed in several chronic tests on laboratory animals. Acute toxicity to human beings has been observed for 2,4,6-TNT. Chronic toxicity to human beings has not been observed for any of the nitroaromatic contaminants. Data indicating adverse effects on aquatic biota are also reviewed.

In Sec. 4.0, the distribution of contaminants and the fate and transport of nitroaromatics are presented for the Acids Area/Yellow Water Reservoir Area, the Red Water Reservoirs Area, and the Pond 13/Wet Well Area. In the Acids Area/Yellow Water Reservoir Area, three distinct contaminant sources have been identified:

1. Contaminated surficial soil adjacent to the yellow water neutralization facility,
2. The yellow water sewerline, which extends from the neutralization facility to the outfall in the Mill Creek Tributary, and
3. The sediments of the Yellow Water Reservoir.

In the Red Water Reservoirs Area, sources of contamination include the sediments of Ponds 1 and 2 and soils associated with the Red Water sewerline. The sewerline deposits have been addressed as part of the first operable unit (ESE, 1986c) and are discussed herein only to the extent that they contribute to the observed patterns of contamination.

At the Pond 13/Wet Well Area, the primary source of contamination is the subsurface sediments of the wet wells which are associated with the sewerline and former pumping station in this area.

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In all areas of concern in the second operable unit, nitroaromatic contaminants are leached from the sources by infiltrating rain water, thereby contaminating a shallow sand aquifer.

In Sec. 5.0, information on the exposure pathways and the population at risk is presented. At the Acids Area/Yellow Water Reservoir Area, the only significant pathway at present is direct contact with surficial soils. If the area is developed for industrial use in the future, it can be assumed that workers could be exposed to contaminated soils/sediments. Onsite construction or landscaping activities could expose contaminated subsoils.

At the Red Water Reservoirs Area, the ground water/drinking water pathway is the only potential exposure pathway. The data generated during the RI indicated that exposure to contamination from the Red Water Reservoirs does not occur at this time. The potential future exposure levels for hypothetical new well installation near the McClintic Wildlife Station boundary was based on 2,4-DNT levels observed at Well GW47.

At Pond 13, the exposure pathway of concern results from leachate generation by contaminated sediments in the wet wells, ground water transport in the shallow sand aquifer, and discharge via a small seep to Pond 13, where aquatic biota are exposed. Since Pond 13 is currently closed to fishing, aquatic biota are the only population exposed. If Pond 13 were reopened, fishermen might be exposed after bioaccumulation of contaminants from water or sediments.

In Sec. 6.0, the toxicity and exposure information are combined to determine the maximum safe residual contaminant levels in water, soil, and sediment. Comparison of observed levels with these criteria provides appropriate action levels for the feasibility study.

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Criteria have been developed in consideration of all realistic exposure pathways by which people or wildlife may be exposed to the contaminants, including direct exposure, uptake through the food chain, and consumption of water. Criteria development was modeled on the Preliminary Pollutant Limit Value (PPLV) methodology developed at U.S. Army Medical Bioengineering Research and Development Laboratory (USAMBRDL). The PPLV methodology represents an approach to criteria development based on site-specific exposure and risk assessment techniques.

The EA analyses are based on the following set of objectives for remediation:

1. Upon completion of remediation, McClintic Wildlife Station will be managed as a hunting and fishing area with free access; recreational users should not be exposed directly or through game and fish to contamination levels that pose a significant risk.
2. Ground water at the McClintic Wildlife Station boundary and offsite should be safe to drink, implying a negligible individual lifetime cancer risk of 10^{-5} or 10^{-6} for any individual who may tap that ground water supply in the future.

Ground water resources that may have become contaminated are not now used for potable supply. The area is served by a municipal supply, and any new residents can be connected to this supply. However, they are not required to, and the ground water resource is a "usable" aquifer.

Overwhelming institutional constraints militate against the use of ground water on McClintic Wildlife Station as a potable supply. McClintic Wildlife Station is expected to remain a wildlife station in perpetuity. The State of West Virginia has the incentive, intent, and authority (barring national emergency, in which case the Army could take control of the property, presumably for industrial use) to maintain the land in this use.

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These factors provide a context in which Objective No. 2 could be applied with some degree of flexibility. Although it would be desirable to achieve drinkable ground water quality at McClintic Wildlife Station boundaries, endangerment by the potential ingestion of usable water supplies not now used in a locale where an alternative supply is readily available is unlikely and could be prevented by a variety of means, including:

1. Requiring existing or future residents whose supply is threatened to hook up to the available municipal supply, or
2. Expanding the McClintic Wildlife Station to include some or all of the areas within ground water contaminant plumes.

The safe contaminant levels presented here are based on Objectives 1 and 2, and the assumption that McClintic Wildlife Station boundaries remain as they are today.

Criteria based on health risks have been developed for soils/sediments in the Acids Area/Yellow Water Reservoir. Criteria are presented for ground water for areas outside the McClintic Wildlife Station boundary. Independent criteria designed for the protection of both human health and aquatic life have been developed for surface water of the Pond 13/Wet Well Area, which is on the McClintic Wildlife Station. RI data support the hypothesis that contamination associated with the Pond 13/Wet Well Area does not migrate off McClintic Wildlife Station in hazardous concentrations. Exposure scenarios appropriate for the mandated use of McClintic Wildlife Station property as a wildlife management area with unrestricted access are applied to derive criteria for the Pond 13/Wet Well Area.

Contamination in these three areas can affect several distinct populations at risk. The reasonable assumption that these populations are independent is adopted. The populations potentially at risk are:

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1. Potential future workers at a hypothetical industrial facility that may be constructed at the Acids Area/Yellow Water Reservoir,
2. Existing and/or potential future residents downgradient of contaminant plume in ground water associated with the Red Water Reservoirs,
3. Recreational fishermen or other visitors to the Pond 13/Wet Well Area, and
4. Freshwater aquatic biota (Pond 13/Wet Well Area).

The Acids Area/Yellow Water Reservoir is on a parcel of land contiguous with McClintic Wildlife Station and owned by the Mason County Development Authority. The land currently is not used industrially or for any other purpose. Its location, developed infrastructure, and zoning suggest that future industrial use is a reasonable assumption. Adjacent commercial, institutional, and industrial development make the land unattractive for residential development. It is assumed that industrial use could occur in the future.

Pertinent to the Red Water Reservoirs, criteria for ground water used as a drinking water supply are presented. These criteria should be met off McClintic Wildlife Station property. The source of ground water contamination in this area is attributed primarily to sediments of the Red Water Reservoirs. These sediments are covered by clean, recent sediments; therefore, there is no potential for direct exposure or exposure through the aquatic food chain. These sediments can contribute to endangerment via only the ground water pathway. As guidance to the FS, criteria have been developed for these sediments such that achieving these criteria would ultimately result in meeting the ground water criteria at the boundary. The only reason for evaluating remedial options addressing the sediments would be to limit the duration of an action involving ground water (i.e., cost-effectiveness). AR 804872
necessary to achieve the sediment criteria to eliminate the endangerment.



In the Pond 13/Wet Well Area, surface water is the environmental medium resulting in exposure to either fishermen or aquatic life. The observed surface water contamination results from ground water discharge, and the ground water contamination in turn appears to be associated with the abandoned Wet Well sediments. Thus, complementary and self-consistent criteria have been developed for the Wet Well sediments, ground water, and surface waters with the objective of effecting acceptable surface water concentrations. Exposure pathways are completed via only the surface water medium; therefore, criteria for Wet Well sediments and ground water are provided as guidance to the FS to support evaluation of cost-effective options. Contamination of sediments and ground water in the Pond 13/Wet Well Area does not result directly in an endangerment. The contaminated sediments are subsurface, and the ground water is not used.

In all cases, remedial alternatives that reduce exposure cost-effectively and interdict contaminant migration at various points in the pathways should be evaluated. For example, ground water remediation at the McClintic boundary downgradient of the Red Water Reservoirs could be evaluated with or without remediation of contaminated sediments in the reservoirs. Cost-effectiveness of the alternative would be the principal basis for choosing between two such alternatives since either would remedy the endangerment. Thus, remedial objectives are to ensure that:

1. Ground water used as a drinking water supply will be safe to drink,
2. Soils in the Acids Area/Yellow Water Reservoir will not result in unacceptable exposure to future workers on the site, and
3. Surface waters on McClintic Wildlife Station will not be toxic to aquatic life or result in unacceptable exposure to fishermen.

These objectives would be met if the numerical criteria in Table 1 are achieved. Corollary criteria for other media developed in AR 304-673 subsidiary to the criteria in Table 1 because they are designed to further compliance with these objectives.



Table 1. Remedial Objectives for Second Operable Unit

Compound	Ground Water Used as Drinking Water Supply (µg/L)	Soils, Industrial Land Use (mg/kg)	Surface Waters, McClintic (µg/L)
TNT	50	4,000	60
DNB	14	1,200	160
TNB	200	18,000	80
2,4-DNT			
10^{-6} Risk	0.11*	10	3.4
10^{-5} Risk	1.1	100	34
2,6-DNT			
10^{-6} Risk	0.022*	2	0.67*
10^{-5} Risk	0.22*	20	6.7
Total Nitroaromatics			
10^{-6} Risk	NA	200	NA
10^{-5} Risk		2,000	

*Below detection limit.

Source: ESE, 1986a.

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In Sec. 7.0 of this report, health and environmental risks associated with exposure to site contamination are evaluated. Risk estimates are presented for existing populations and current conditions, as well as hypothetical future land or water uses by populations not currently at risk. In general, no population is currently being exposed to unacceptable levels of contamination, but future populations might be at risk if land or water use changed and no remedial actions were taken. In consideration of this, the No-Action Alternative is not viable in any area included in the second operable unit.

In the Acids Area/Yellow Water Reservoir, the carcinogenic risk resulting from occasional exposure to contaminated surficial soils under the existing land use is estimated to be less than 1.8×10^{-6} . If the area was developed industrially, the most probable future land use, workers could be exposed to DNT concentrations resulting in a 2.8×10^{-5} individual cancer risk. Furthermore, exposure to 2,4,6-TNT, a noncarcinogenic contaminant, would exceed recommended exposure levels by a factor of 2.7. Surficial soil contamination in this area has resulted in adverse environmental impacts on vegetation as indicated by the lack of vegetation in a contaminated hot spot.

Levels of contamination in the surficial water table aquifer in the Acids Area/Yellow Water Reservoir exceed acceptable levels for drinking water. The shallow water table aquifer is not used as a potable supply anywhere in the vicinity of WVOW since the preferred ground water supply is the deeper sand and gravel aquifer. Contamination of this aquifer theoretically represents an endangerment since there are no prohibitions or feasibility constraints against its use. As a practical matter, however, this contamination poses no risk to public health since there is no reason to believe that it would be used in the future, given the traditional use of the deeper aquifer. The contaminated portion of the surficial aquifer is in an area developed for institutional and industrial use, which is unattractive for residential development.

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In the Red Water Reservoir Area, ground water used as a source of domestic supply is contaminated. Existing wells exhibit acceptable levels of contamination and the ground water quality has apparently reached a steady-state (maximum) concentration, so existing water users are not at risk now or in the future. If new wells were installed in areas outside but near McClintic Wildlife Station, these wells might be contaminated to unacceptable levels. The carcinogenic risk associated with exposure to the DNTs for a hypothetical new domestic well in the center of the plume at the McClintic Wildlife Station boundary could be as great as 4.8×10^{-4} , which is outside the acceptable risk range for CERCLA response actions. Risks due to noncarcinogenic contaminants and risks to nonhuman biota associated with the Red Water Reservoirs Area are negligible.

Existing populations are not exposed to unacceptable levels of contamination in the Pond 13/Wet Well Area because Pond 13 is closed to fishing. If no remedial action were taken and Pond 13 were reopened for fishing, bass fishermen and others with whom they shared their catch could experience a cancer risk as high as 1.7×10^{-5} due to exposure to the DNTs. Exposure to noncarcinogenic contaminants and risks to aquatic biota are within acceptable limits in this area.

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1.0 INTRODUCTION

The U.S. Army Toxic and Hazardous Materials Agency (USATHAMA) issued Contract No. DAAK11-83-D-0007 to Environmental Science and Engineering, Inc. (ESE) to perform tasks relating to the Multi-Installation Eastern Sites Environmental Contamination Surveys Program. Task Number 0004, Delivery Order Number 0005, comprises a remedial investigation/feasibility study (RI/FS) of the former West Virginia Ordnance Works (WVOW) to assess contamination and contamination migration in the soils, surface water, and ground water as a result of past 2,4,6-trinitrotoluene (2,4,6-TNT) manufacturing operations and to evaluate any necessary remedial action alternatives.

This endangerment assessment (EA) report is a component of the feasibility study (FS). The purpose of the EA is to summarize and interpret remedial investigation (RI) data, identify contaminant transport pathways and receptors, and assess actual or potential harm to public health, welfare, or the environment from hazardous materials originating on WVOW. The EA provides a transition from the RI (ESE, 1986d) to the FS which is in progress. In essence, it justifies the need for remedial action and serves to focus remedial action alternatives.



1.1 SITE DESCRIPTION AND HISTORY

The WVOW site covers approximately 8,323 acres in Mason County, WV. It is approximately 58 miles (mi) northwest of Charleston, 41 mi northeast of Huntington, and 6 mi north of Point Pleasant, WV, on the east bank of the Ohio River. Approximately one-third of the area is currently occupied by the Clifton F. McClintic State Wildlife Station (McClintic Wildlife Station), operated by the State of West Virginia Department of Natural Resources (DNR), which is 2,788 acres in size.

From 1942 to 1945, WVOW operated to produce trinitrotoluene (TNT) explosive. Production of this material during World War II (WWII) resulted in contamination of the soils of the industrial area.

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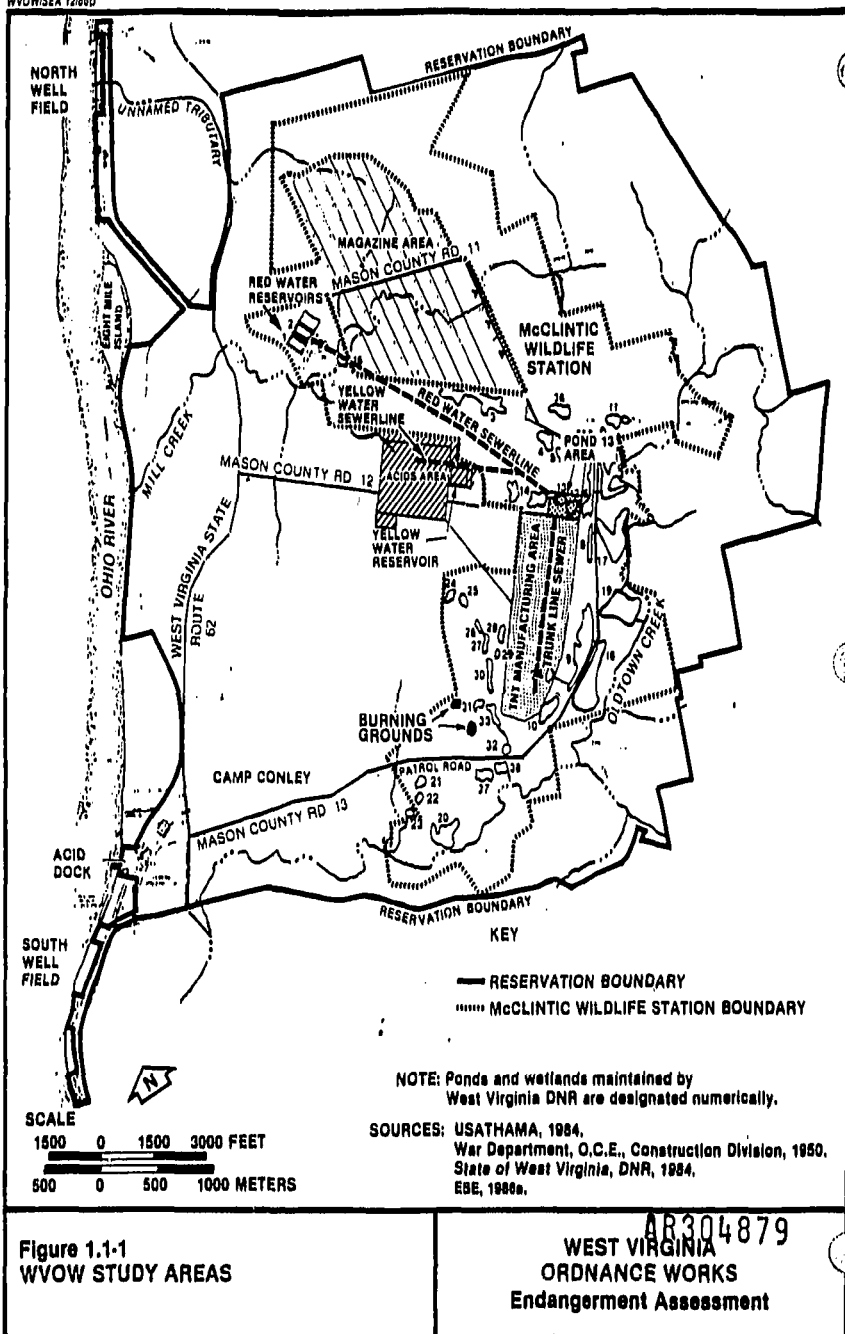
facilities, and industrial wastewater disposal facilities by TNT and associated byproducts and environmental transformation products. TNT was shipped to various Government installations to be loaded into munitions or for other uses. No loading of munitions or testing of ordnance was conducted at WVOW.

At the close of operations in 1945, WVOW was decontaminated to place it in standby status. Later in 1945 the plant was declared surplus and the facilities salvaged or disposed of. No records currently exist regarding the general extent of this decontamination. Because the industrial area was contaminated to the extent that complete decontamination was not feasible, a portion of the land was not released to private ownership but was transferred to the state of West Virginia for wildlife conservation. This area of the former WVOW, including most of the former industrial area, forms the McClintic Wildlife Station. Smaller portions of the nonindustrial areas of the site were excecised by the Government and are now owned by Mason County, WV, or by private owners. The layout of the former industrial facility and the present boundary of McClintic Wildlife Station are shown in Fig. 1.1-1.

In May 1981, a seepage of red water was observed adjacent to Pond 13 located on the McClintic Wildlife Station. This pond was located near the former TNT wastewater trunk sewerlines and pumping station. This incident was investigated by the West Virginia DNR and U.S. Environmental Protection Agency (EPA). The shallow ground water discharging to Pond 13 was found to be contaminated by 2,4-dinitrotoluene (2,4-DNT) and 2,6-dinitrotoluene (2,6-DNT) [up to 7,100 micrograms per liter (µg/L) of 2,4-DNT and 1,300 µg/L of 2,6-DNT], 2,4,6-TNT (166 µg/L in one sample), and phenol (31 µg/L).

Based on these and other studies by the West Virginia DNR and by EPA contractors in 1981 and 1982, WVOW has been ranked as the 84th site on

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the National Priorities List (NPL) under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980 (PL 96-510 amended by PL 97-272).

1.2 SUMMARY OF CONTAMINATION STATUS

The following paragraphs are a summary of the overall site contaminant sources and contaminated media. This summary is based on site characterization and sampling and analysis results reported in two RI reports (ESE, 1986d; ESE 1987b). Sampling and analysis were conducted in accordance with ESE's USATHAMA-approved quality assurance (QA) plan (ESE, 1984b). This QA plan has been reviewed and accepted by EPA. Any data developed using procedures not in substantial compliance with this QA plan were not reported. The data are considered to be reliable. Wells installed during the RI survey were constructed according to USATHAMA monitor well specifications, and samples from these wells are expected to be representative of ground water quality in the screened interval. For comparison with initial EPA investigations of the site, samples were collected from several wells in the Pond 13/Wet Well Area that were not constructed in compliance with USATHAMA monitor well specifications. Samples taken from these wells, denoted with the prefix EPA, may not be representative of ground water quality. Locations of the wells sampled during the RI are shown in Fig. 1.2-1.

The principal sitewide contaminants are nitroaromatic residues, and the predominant compound observed was 2,4,6-TNT; 1,3,5-trinitrobenzene (1,3,5-TNB) and 2,4-DNT were also widely distributed. The major nitroaromatic contaminant source areas are shown in Fig. 1.2-2 and include:

1. The surface and subsurface soils in the TNT Manufacturing Area,
2. The industrial sewerlines in the TNT Manufacturing Area and the trunk sewerlines leading from the Pond 13 area to the outfalls, and
3. The surface soils in the East and West Burning Grounds.

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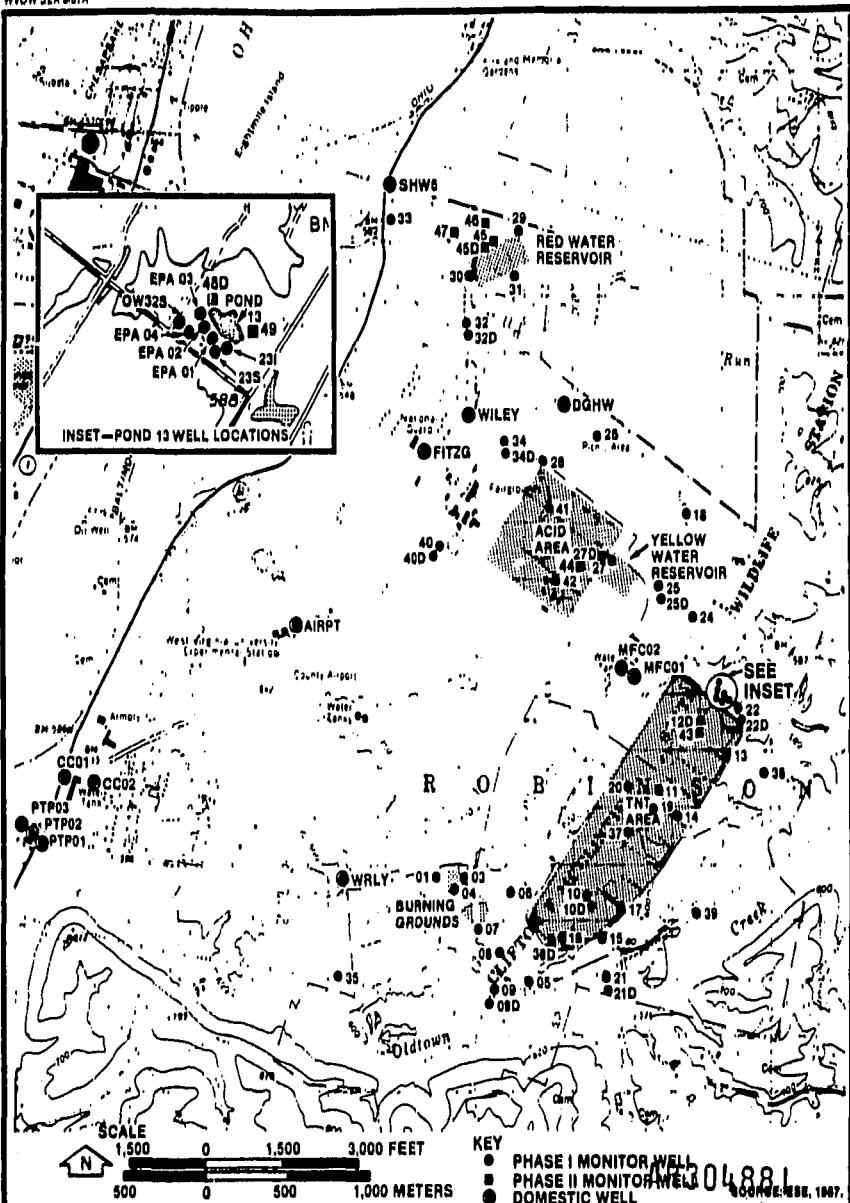
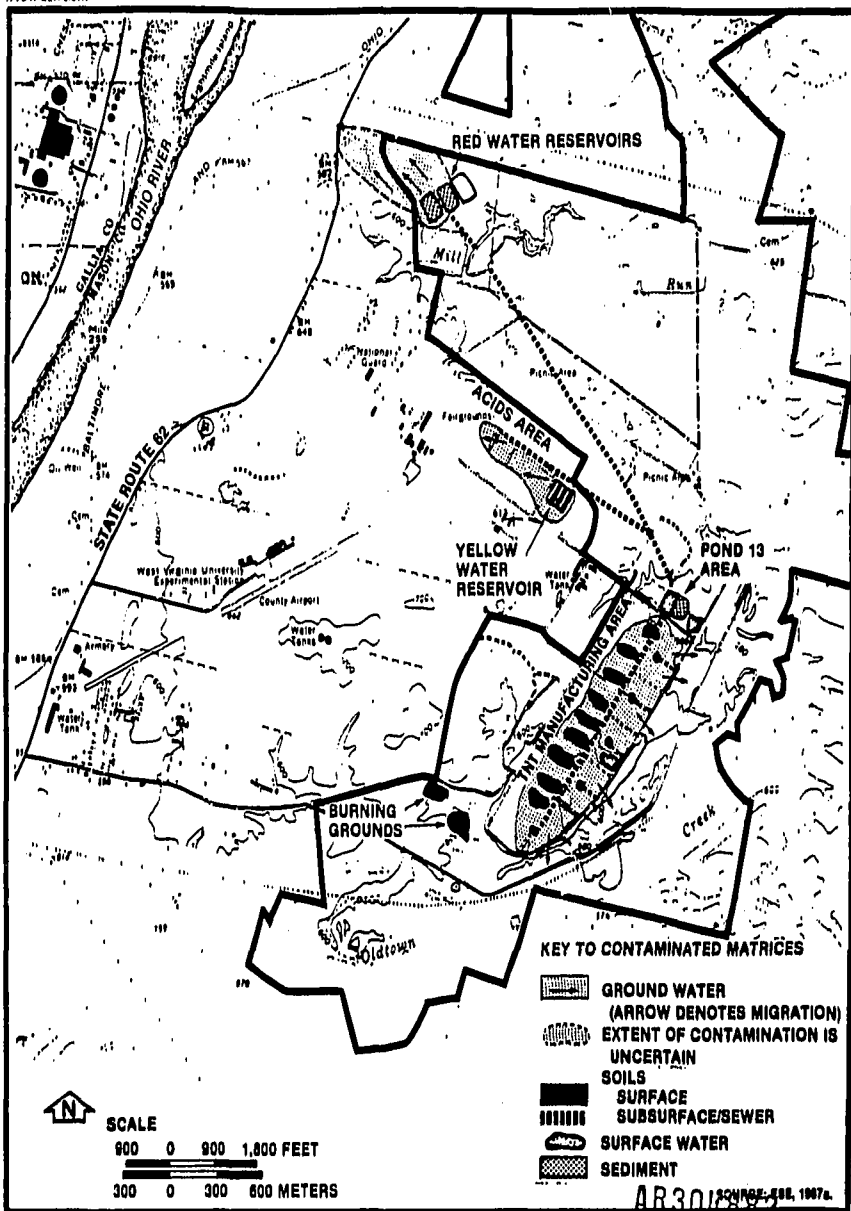


Figure 1.2-1
LOCATIONS OF MONITOR WELLS, EPA WELLS,
AND OFFSITE DOMESTIC WELLS

WEST VIRGINIA
ORDNANCE WORKS
Endangerment Assessment

WVOW SEA 807A





In these areas concentrations ranging to the low percent levels (<10 percent) were encountered. Fist-sized pieces of crystalline nitroaromatic residue were encountered in the Burning Grounds Area. These source areas contribute surface water and ground water contamination by nitroaromatics and represent a hazard to human beings and wildlife as a result of direct contact. A very small area [approximately 100 square feet (ft²)] of surface soils in the Acids Area/Yellow Water Reservoir area is also contaminated with nitroaromatics to approximately the 1-percent level.

The Red Water Reservoirs sediments, Yellow Water Reservoir soils, and the Wet Well/Pond 13 seep area sediments are contaminated to a lesser degree and do not represent a direct contact threat, but contribute to surface water and ground water contamination. These source areas are shown in Fig. 1.2-2.

Asbestos, disposed primarily at the West Burning Grounds, represents a direct contact hazard in this area. Surface water migration of asbestos is occurring into the drainage leading from this source area into Oldtown Creek. Waters of the drainage area and Oldtown Creek are not used as drinking water sources or for body contact recreation. Polynuclear aromatic hydrocarbons (PAHs) and lead also were observed in the West Burning Grounds. Although these contaminants represent a potential contact hazard in the source area, no generalized migration appears to have occurred. Asbestos also exists offsite in the powerhouses and Mason Furniture Co. Access to the south powerhouse is not restricted. The Mason Furniture Bldg. is privately owned and currently is occupied by an injection molding company.

Table 1.2-1 indicates the maximum concentrations observed in various contaminated areas and media for categories of contaminants associated with the WVOW site. In each area, sampling strategy was designed to identify and sample the most contaminated areas, so the maximum

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Table 1.2-1. Summary of Contamination Status of the Second Operable Unit at WVOW

Environmental Medium	Contaminant	Maximum Concentration Detected*
<u>Acids Area/Yellow Water Reservoir</u>		
Soils	Nitroaromatics	1% (isolated area approximately 100 ft ²)
	Lead	100 µg/g
Sewerlines	Nitroaromatics	2,830 µg/g
Surface Water	Uncontaminated	--
Ground Water	Nitroaromatics	15 µg/L
<u>Red Water Reservoirs</u>		
Soils	Uncontaminated	--
Sewerlines	Nitroaromatics	0.2%
Surface Water	Uncontaminated	--
Sediments	Nitroaromatics	2,210 µg/g
Ground Water	Nitroaromatics	17 µg/L
<u>Pond 13/Wet Well Area</u>		
Soils	Nitroaromatics	5.5 µg/g
Sewerlines	Uncontaminated	--
Surface Water	Nitroaromatics	70 µg/L
	Lead	32 µg/L†

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Table 1.2-1. Summary of Contamination Status of the Second Operable Unit
at WVOW (Continued, Page 2 of 2)

Environmental Medium	Contaminant	Maximum Concentration Detected*
<u>Pond 13/Wet Well Area (Continued)</u>		
Sediments	Nitroaromatics	4,240 µg/g
Ground Water	Nitroaromatics	50,000 µg/L

Note: µg/g = micrograms per gram.

*In each area, sampling strategy was designed to identify and sample the most contaminated areas, so the maximum concentrations are not necessarily representative of the typical concentration in each source area. In virtually all source areas, several samples were collected that were uncontaminated thus realistically defining the extent of contamination.

†Lead in ground and surface waters did not exceed relevant standards or criteria [e.g., National Interim Primary Drinking Water Regulations (NIPDWR) = 50 µg/L].

Source: ESE, 1987a.

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concentrations are not necessarily representative of the typical concentration in each source area. In virtually all source areas, several samples were collected that were uncontaminated, thus realistically defining the extent of contamination. Details of the contaminant distribution in each source area are presented in the RI report (ESE, 1986d).

ESE (1986b) summarized sampling and analysis of potable water supply wells in the vicinity of WVOW. Samples collected in 1985 indicated that all potable water supply wells were free of contaminants characteristic of the site. A sample collected in April 1986 from the Schwartz well (near SR 62, approximately 750 meters (m) west-northwest of the Red Water Reservoirs) contained 0.9 µg/L of 1,3-dinitrobenzene (1,3-DNB) and 0.2 µg/L of 2,4,6-TNT.

Reinvestigation of analytical data from the 1985 sampling of the Schwartz well indicated trace levels of nitroaromatic contamination below the verifiable detection limit, based on documented quality assurance/quality control (QA/QC) criteria. The low levels observed in the Schwartz well are well below allowable drinking water levels (see Sec. 6.3). The Schwartz well is not in use for potable supply because the residence has been connected to the municipal water supply.

1.3 AREAS ADDRESSED IN THIS REPORT

The initial RI report (ESE, 1986d) detailed the contamination status of WVOW. For certain areas (specifically the Acids Area/Yellow Water Reservoir, Red Water Reservoirs, and Pond 13/Wet Well Area, hereafter termed the second operable unit), important uncertainties remained regarding the source, extent, and migration potential of contaminated ground water. These uncertainties dictated that additional field data be collected to support informed remedial decisionmaking. This effort was completed and reported by ESE in a supplemental RI report (1987b). In the other areas (the TNT Manufacturing Area, Burning Grounds, and Industrial Sewerlines) the EA and FS have been completed (ESE, 1986b; and ESE, 1986c, respectively). This report completes EA activities for the WVOW site and addresses the second operable unit.



2.0 ENVIRONMENTAL FATE PROPERTIES OF SITE CONTAMINANTS

2.1 NITROAROMATIC COMPOUNDS

The physicochemical properties of nitroaromatic compounds found at WVOW are summarized in Table 2.1-1. The nitroaromatics are similar to each other with respect to solubility and properties affecting adsorption. They differ in volatility in that the dinitrotoluenes are substantially more volatile than the other nitroaromatic compounds.

2.1.1 Sorption

The tendency of 2,4,6-TNT to adsorb to soils characteristic of contaminated areas of WVOW was studied experimentally by ESE (1985), who collected samples from three soil series characteristic of WVOW: the Lakin loamy, fine sand, which is found in the Red Water Reservoir area; the Chilo sandy loam, which is found in the Acids Area/Yellow Water Reservoir area and the TNT Manufacturing Area; and soil of the Markland McGary complex which is characteristic of the Burning Grounds Area. These soils were characterized (see Table 2.1-2) in terms of their organic carbon content (which ranged from 0.1 to 1.2 percent), clay content (approximately 5 to 38 percent), cation exchange capacity (CEC) [2.9 to 22 milliequivalents per 100 grams (meq/100 g)], and pH (5.8 to 6.4). Subsequently, the tendency of 2,4,6-TNT to adsorb to these soils was determined experimentally. These experiments were performed in the context of soil adsorption studies of six explosive compounds on 12 soils from across the country, which indicated that TNT adsorption was significantly correlated with soil organic carbon and cation exchange capacity.



These data are much more specific and definitive than the environmental fate information usually available for hazardous waste site investigations. Nonetheless, there are still significant uncertainties regarding the adsorption phenomenon, as well as other environmental fate processes at this site. For example, the soil characteristics affecting adsorption, particularly the organic carbon content, un

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Table 2.1-1. Reported Physicochemical Properties of Nitroaromatic Compounds Found at WOM

Compound	Molecular Weight	Vapor Pressure (torr)	Octanol/Water Partition Coefficient (log K_{ow})	Solubility (mg/L)	Henry's Law Constant (torr/M)	Sorption Partition Coefficient (K_{oc})
2,4,6-TNT	227	10^{-4}	1.9	117 to 200	0.18	319 (190-534)
2,4-DNT	182	5.1×10^{-3}	2.0	270	3.4	132 (87-201)
2,6-DNT	182	0.018	—	180	18	158 (100-269)
1,3,5-TNB	213	2.2×10^{-4}	1.2	35	1.3	303 (177-520)
1,3-DNB	170	0.039	—	360	3.2	64

Notes: Ranges denote values estimated or measured by different investigators.

mg/L = milligrams per liter.

torr/M = torr per molar.

Vapor pressure, solubility, and Henry's Law Constant as presented at 20°C where available.

Source: Adapted from ESE, 1985.

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Table 2.1-2. Characteristics of Some WVON Soil Series and Their Tendency to Adsorb 2,4,6-TNT

Soil Series (as identified in ESE, 1985)	Organic Carbon Content (Z)	Clay Content (Z)	CEC (meq/100 g)	pH	Adsorption Coefficient (K _d)
Lakin, Surface Horizon (WV-1)	1.2	10.7	8.7	6.1	1.6
Lakin, 1.2 to 1.6 ft (WV-2)	0.1	4.8	2.9	6.4	1.0
Markland McGary, 3.8 to 4.2 ft (WV-3)	0.2	30.1	22.0	6.3	0.3
Chilo, 2 to 3 ft (Chilo-1)	0.4	37.8	19.0	5.8	1.6

Note: ft = feet.

K_d = concentration in soil ÷ concentration in water, at equilibrium, expressed as mL/g.

Source: Adapted from ESE, 1985.

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over the site and through the soil column. Organic carbon content of surficial soils is expected to range from 0.1 to 1.2 percent, while the carbon content of subsurface soils is expected to range from 0.1 to 0.4 percent. Similarly, other soil characteristics (CEC, pH, and clay content) that apparently affect the adsorption of nitroaromatics (ESE, 1985) also vary across the site. The experimental data of ESE (1985), considered in the context of the attainable precision of estimation presented by Lyman et al. (1982) for the relevant methods, indicate that the adsorption of nitroaromatic compounds to uncharacterized WVOW soils can be estimated to range approximately as shown in Table 2.1-3.

2.1.2 Transformation/Degradation

The fact that nitroaromatic compounds are still observed in soils and ground waters at WVOW some 40 years after cessation of explosives production indicates that these compounds are relatively resistant to chemical and microbial degradation under environmental conditions. 2,4,6-TNT is the dominant nitroaromatic contaminant observed at WVOW and other munitions sites, whereas the dinitrotoluenes (DNTs) are believed to be the most toxic and are carcinogenic in animals. Because of the predominance of 2,4,6-TNT and the toxicity of the DNTs, these contaminants have been studied more extensively than the remaining nitroaromatics and environmental transformation processes involving them are emphasized here in relation to the better data base.

Photolysis is the dominant transformation process affecting nitroaromatics in surface environmental systems that receive direct solar radiation (Spanggord et al., 1980b). Photolysis rates are estimated to be 10 to 1,000 times greater than biotransformation rates in natural waters. All other transformation processes are insignificant in comparison to photolysis. Photolysis of 2,4,6-TNT predominantly yields 1,3,5-TNB and 2,4,6-trinitrobenzaldehyde (Spanggord et al., 1980b). In an experiment reviewed by Spanggord et al. (1980b), trinitrobenzaldehyde in natural surface water was rapidly converted to trinitrobenzene (TNB).
AN 304890

Table 2.1-3. Representative Values of the Adsorption Coefficient

Compound	Adsorption Coefficient* (K _d)			
	Soils		Subsoils (Below 1 ft)	
	Best Estimate	Probable Range	Best Estimate	Probable Range
2,4,6-TNT	0.9	0.3 to 2.7	0.8	0.3 to 2.1
2,4-DNT	0.4	0.1 to 1.9	0.3	0.1 to 1.4
2,6-DNT	0.4	0.1 to 1.9	0.3	0.1 to 1.4
1,3,5-TNB	0.9	0.2 to 4.3	0.8	0.2 to 3.8
1,3-DNB	0.2	0.1 to 1.0	0.2	0.1 to 0.7

*Adsorption coefficient, K_d (mL/g) = C_s/C_l

where: C_s = concentration adsorbed to soil solids based on dry weight of solids (µg/g), and
C_l = concentration in the soil solution (µg/mL).

Source: Adapted from ESE, 1985.

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The toxicity of TNB is similar to that of 2,4,6-TNT [i.e., acute LD50s range from 505 milligrams per kilogram (mg/kg) for TNB to 1,320 mg/kg for TNT; both have been found to be mutagenic, but neither has tested positive for carcinogenicity], whereas the toxicity of other potential transformation products is not well known. 1,3,5-TNB is, however, substantially more volatile than 2,4,6-TNT and will volatilize from surface water (Spanggord et al., 1980b) and surficial soil systems (ESE, 1985). DNTs and nitrobenzenes are all rapidly photolyzed and are substantially volatile (Spanggord et al., 1980a). Photolysis coupled with volatilization probably plays the most significant role in removing nitroaromatic compounds and their transformation products from ecosystems at the earth's surface.

In subsurface soils and ground water, biotransformation may be the most important transformation process affecting 2,4,6-TNT. Transformation products include amino-dinitrotoluenes, diamino-nitrotoluenes, hydroxylamino-dinitrotoluenes, tetranitro-azoxytoluenes, and possibly di- and tri-nitrobenzaldehydes (Roy F. Weston, Inc., 1985). Although little toxicological information is available on these metabolites, there is no evidence that they are more toxic than 2,4,6-TNT. Historical literature (Spanggord et al., 1980a) suggested that DNTs formed from 2,4,6-TNT due to microbial transformation in the dark. The most recent available evidence indicates more strongly that the animal carcinogens 2,4-DNT and 2,6-DNT are not transformation products of TNT under environmentally relevant circumstances (Roy F. Weston, Inc., 1985).

The relative importance of various transformation processes is very similar for the DNTs as well as for the other nitroaromatics. Transformation products are also similar, with reduction of a nitro group to an amino group as the most commonly observed transformation route. None of the many studies reviewed by Roy F. Weston, Inc. (1985) clearly indicated mineralization/degradation to innocuous end products.

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2.2 LEAD

Lead is adsorbed readily by soils. Adsorption capacity of soils for lead increases with pH, CEC, organic carbon content, and phosphorus levels (Perwak et al., 1980). Lead is readily adsorbed above a soil pH of 5. Soils representative of contaminated soils at WVOW were collected and characterized as described in Sec. 2.1.1.1 (ESE, 1985). Based on information presented by Perwak et al. (1980), these soils will have the capacity to sorb at least 7,000 milligrams (mg) of lead per kilogram of soil. In several studies summarized by Perwak et al. (1980), only 0.6 to 1.6 percent of the total lead in soils should be leachable. Consequently, runoff of suspended particles with adsorbed lead, rather than leaching of soluble lead, is expected to be the dominant pathway for lead migration from contaminated soils.



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3.0 TOXICITY ASSESSMENT

The toxicity assessment, as a component of the EA process, describes and evaluates the health and environmental hazards associated with specific contaminants. Based on the RI results as summarized in Table 1.2-1, the contaminants of concern for the second operable unit are: 2,4,6-TNT; 2,4-DNT; 2,6-DNT; 1,3,5-TNB; 1,3-DNB; and lead. A toxicological assessment for WVOW contaminants was conducted to qualitatively evaluate the available data on pharmacokinetics, human health effects, and effects on fish and wildlife. A dose-response assessment was then performed to quantify risks associated with exposure to each contaminant and to estimate acceptable intake levels. Acceptable levels for noncarcinogens are presented as acceptable daily intake (ADI) values; for carcinogens, cancer potency factors are calculated based on mathematical models or dose-response data.

3.1 2,4,6-TRINITROTOLUENE (2,4,6-TNT)

3.1.1 Pharmacokinetics

2,4,6-TNT may enter the mammalian body through any of three primary routes of absorption: the gastrointestinal tract (GI), the skin, or the lungs. During exposure to 2,4,6-TNT, the powder may be ingested by mouth and gain access to the stomach and gastrointestinal tract where it is absorbed. Skin absorption in humans and laboratory animals has been demonstrated by recovery of 2,4,6-TNT metabolites in urine (Voegtlin *et al.*, 1921; Neal *et al.*, 1944; Haythorn, 1920). A recent study by El-hawari *et al.* (1981) found that radioactive-labeled 2,4,6-TNT administered orally to rats, mice, rabbits, and dogs was readily absorbed and excreted mainly in the urine and, to a lesser degree, in feces. Dermal absorption was highest in rabbits (68 percent of dose recovered after 24 hours), followed by mice (42 percent), rats (23 percent), and dogs (17 percent). High levels of radioactivity were found in blood, liver, kidneys, and occasionally spleen and lungs. In the same study, 2,4,6-TNT was absorbed by each of the species after dermal application, and most 2,4,6-TNT was eliminated in urine. In these studies, 2,4,6-TNT

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was administered in a peanut oil mixture. This mode of administration may affect absorption rates. The extent of absorption could not be accurately determined, however, since radioactivity measured in the feces and GI tracts of laboratory animals includes dose fractions that are absorbed, excreted in bile, and reabsorbed by the intestines. In dogs, 99 percent of the oral dose was recovered in the urine, feces, GI tract, blood, and tissues. The recovery efficiencies for other species were 96 percent in rabbits, 94 percent in mice, and 92 percent in rats. Total amount of 2,4,6-TNT excreted in urine and feces 24 hours following dermal application was less than the amount excreted following oral administration of the same dose. 2,4,6-TNT was found to be metabolized extensively in all species, and by all routes of administration. The metabolic products were mainly reduction derivatives, including the hydroxylamines, the monoaminodinitro- and the diaminomononitro-derivatives and azoxy-coupled compounds.

Experimental evidence indicates that in small doses 2,4,6-TNT is rapidly detoxified by the liver and the metabolic products are excreted by the kidneys. Lee et al. (1975) studied the distribution and excretion of radioisotope-labeled 2,4,6-TNT following oral administration in rats. The researchers found that approximately 60 to 74 percent of an oral (nonlethal) dose was absorbed in 24 hours. Within 24 hours after dosing, an average of 58.6 percent of the dose was excreted in urine. The gastrointestinal tract and the feces were found to contain 36 percent of the radioactivity. Small amounts of radioactivity were also found in blood, liver, kidney, and skeletal muscles.

Extensive work has been carried out to identify 2,4,6-TNT metabolites in humans and experimental animals. Early studies (Voegtlin et al., 1919; Dale, 1921) found the reduction products (4-amino-2,6-dinitrotoluene and 2,6,2'6'-tetranitro-4,4'-azoxytoluene) excreted in the urine of workers exposed to 2,4,6-TNT. Channon et al. (1944) observed the reduction of a single nitro group of 2,4,6-TNT leading to the formation of 4-amin-

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6-amino-dinitrotoluenes. These researchers also had evidence to suggest that aryl hydroxylamines are produced as a step in the reduction of the nitro group. Such hydroxylamine derivatives have been shown to be more toxic than the parent compound 2,4,6-TNT (Wyon, 1921). In addition to being strong producers of methemoglobin, aryl hydroxylamine intermediates have been implicated in the carcinogenicity responses induced by several carcinogenic amino and nitro compounds (Weisburger and Weisburger, 1973). As discussed in Sec. 3.1.2, however, there is no evidence of 2,4,6-TNT carcinogenicity in humans or laboratory animals.

In vitro experiments suggest that the liver is a major site for 2,4,6-TNT biotransformation (Bueding and Jolliffe, 1946). Studies using liver, muscle, and heart preparations showed that 2,4,6-TNT was reduced by liver homogenates to 4-amino-2,6-dinitrotoluene.

3.1.2 Human Health Effects

The primary site of effect for 2,4,6-TNT is the blood system, although liver degeneration has also been observed in humans. 2,4,6-TNT can cause toxic hepatitis, aplastic anemia, and methemoglobinemia. Animal studies have provided limited evidence of carcinogenicity, and no information was found in the literature for mutagenicity or teratogenicity. The most serious systemic effects of acute 2,4,6-TNT poisoning in humans are toxic jaundice and toxic hepatitis, which may result in liver atrophy and hyperplasia of the bone marrow leading to aplastic anemia.

Several epidemiological studies have been conducted on populations of exposed munitions workers. Cone (1944) and Brill et al. (1944) found that workers in a munitions plant who inhaled an average of 2 micrograms per cubic meter ($\mu\text{g}/\text{m}^3$) of 2,4,6-TNT daily in 8-hour daily work shifts for 2 years experienced transitory leukocytosis with moderate eosinophilia which did not seem to affect overall worker health.

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In a chronic exposure study of 2,4,6-TNT workers, Soboleva (1969) found occupational cataract formation, pathological changes in the peripheral blood, neurasthenia, and polyneuritis. Zakharova and Manoilova (1971) examined 360 workers exposed occupationally to 2,4,6-TNT for 5 years or more. They found eye lesions that developed into cataracts in 45.3 percent of the cases, chronic gastritis, and mild forms of hepatitis. Ermakov et al. (1969) observed the long-term effects of exposure to 2,4,6-TNT at levels in excess of 0.001 mg/L in air. Symptoms of chronic intoxication including lesions of the central nervous and vascular systems were found in 122 of the 574 persons chronically exposed.

McConnell and Flinn (1946) conducted a study of the 22 cases of fatal TNT poisoning that occurred during WWII. Eight of the persons died of toxic hepatitis, 13 died of aplastic anemia, and 1 died from a combination of the two conditions. One-third of the fatalities had been exposed to average 2,4,6-TNT concentrations exceeding the Occupational Safety and Health Administration (OSHA) maximum allowable concentration of 1.5 milligrams per cubic meter (mg/m^3). The amount of 2,4,6-TNT ingested or absorbed through the skin was not known.

Numerous experimental animal studies have been conducted to evaluate the acute and chronic effects of 2,4,6-TNT exposure. A few of the more recent studies are summarized below. Animals vary considerably in their sensitivity to acute 2,4,6-TNT poisoning. Dilley et al. (1982) examined the short-term oral toxicity of 2,4,6-TNT in rats, mice, and dogs. Single dose LD50s were 1,320 and 794 mg/kg in male and female rats, respectively, and 660 mg/kg in both male and female mice. Multiple-dose studies were conducted in all three species for a period of 13 weeks. "No observable effect" levels for 2,4,6-TNT were as follows: dogs, 0.20 milligram per kilogram per day ($\text{mg}/\text{kg}/\text{day}$); rats, 1.42 $\text{mg}/\text{kg}/\text{day}$; and mice, 7.76 $\text{mg}/\text{kg}/\text{day}$. At the highest dose levels, all animals exhibited anemia with reduced erythrocytes, hemoglobin, and hematocrit.

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Abnormal organ weights and body weight alterations were also observed. Reduced testes size was observed in rats at the highest dose regardless of length of exposure. Most toxic effects, except testicular atrophy, were reversible over a 4-week recovery period. Anemia was also observed at intermediate dose levels.

In a subchronic (26-week) oral toxicity study of 2,4,6-TNT in dogs, Levine et al. (1983) observed hemolytic anemia, hepatomegaly, and splenomegaly at doses of 8 and 32 mg/kg/day. The highest dose administered (32 mg/kg/day) was found to be lethal. Liver injury in the form of histologic changes was found at all doses; therefore, a no-observable-effect level could not be derived on the basis of the study.

Levine et al. (1981) also evaluated the subchronic toxicity of 2,4,6-TNT in rats. After 13 weeks, the animals had a weight reduction of approximately 10 percent in males receiving doses of 5 or 25 mg/kg/day, and for females administered 125 mg/kg/day. Other observed effects were reduced food intake, hypercholesterolemia, anemia, splenomegaly, hepatomegaly, and testicular atrophy. Elevated methemoglobin levels and cerebellar lesions were observed only at the 300-mg/kg/day dose level.

Dogs that received subcutaneous doses of 7 mg/kg of 2,4,6-TNT twice a week for 9 months exhibited renal pathological changes (Dyannik et al., 1977). In another study, daily administration of 2,4,6-TNT to dogs and monkeys at less than 1 mg/kg for 90 days showed no toxic effects (Hart, 1974; Martin and Hart, 1974).

Recent studies of the chronic toxicity of 2,4,6-TNT in rats and mice were performed by Furedi et al. (1984a, 1984b). Rats administered TNT for up to 24 months displayed anemia, hepatotoxicity, and urogenital lesions. Hyperplastic and/or neoplastic lesions of the liver, kidneys, and urinary bladder were also observed at doses of 10 mg/kg/day or greater. The observed carcinoma of the urinary bladder indicates that TNT is a

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carcinogen to rats under the experimental conditions, but no dose-response relationship was observed. Based on effects on the spleen, kidneys, and bone marrow at doses of 2 mg/kg/day or above, a no-effect level of 0.4 mg/kg/day was concluded by the study (Furedi, 1984a).

In mice, the major toxic effects observed following dosages up to 70 mg/kg/day for 24 months were anemia and probable hepatotoxicity. Lymphocytosis was seen in animals administered doses of 70 mg/kg/day of TNT and, to a lesser extent, at the 10 mg/kg/day dose level. Leukemia and malignant lymphoma of the spleen were observed only in female mice. The no-effect level determined in this study was 1.5 mg/kg/day (Furedi, 1984b).

3.1.3 Environmental Toxicity

3.1.3.1 Toxicity to Wildlife

No studies concerning 2,4,6-TNT toxicity in wildlife were found in the literature. A number of toxicological studies have been conducted, however, with laboratory animals that may serve as models for wildlife species. These studies have indicated that, in small doses, 2,4,6-TNT is rapidly detoxified and excreted. There is no evidence of mutagenicity or teratogenicity for 2,4,6-TNT based on animal studies. A carcinogenic response was observed in rats by Furedi et al. (1984a), but without a consistent dose-response relationship.

In short-term toxicity studies by Dilley et al. (1982), no-observable-effect levels were reported for dogs (0.20 mg/kg/day), rats (1.42 mg/kg/day), and mice (7.76 mg/kg/day). Effects noted at higher doses included anemia and hematologic changes, abnormal organ and body weights, and testicular atrophy.

Levine et al. (1981) noted similar effects in male rats fed doses of 5 or 25 mg/kg/day and female rats fed 125 mg/kg/day. Dogs and monkeys exposed to less than 1 mg/kg/day for 90 days showed no toxic effects (Hart, 1974;

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Martin and Hart, 1974); however, dogs dosed with 7 mg/kg of 2,4,6-TNT twice a week for 9 months showed renal pathological changes (Dyannik et al., 1977). Kleiner (1969, 1971, 1972) has reported gastric secretory disorders in dogs exposed to 2,4,6-TNT dosages as low as 0.1 mg/kg for periods from 1.5 to 2.5 years.

Numerous earlier studies were conducted to characterize the types of toxic effects and lethality of 2,4,6-TNT. These studies are summarized by Dacre and Rosenblatt (1974).

3.1.3.2 Aquatic Toxicity

2,4,6-TNT photolyzes readily under alkaline conditions in water, although more breakdown products are formed under acidic conditions. Dacre (1980) reported that the majority of toxicity is attributable to the nonpolar, organic soluble fraction of photolyzed 2,4,6-TNT. Results of acute bioassays using fathead minnows indicated an LC50 of 1.9 mg/L for 2,4,6-TNT, whereas LC50s of four phototransformation products ranged from 0.17 to 1.3 mg/L. Daphnia magna were also less sensitive to 2,4,6-TNT than to phototransformation products (6.6 mg/L EC50 for 2,4,6-TNT; 0.34 to 2.7 mg/L EC50 for four product compounds). Fish appear to be more sensitive to 2,4,6-TNT than invertebrates based on static acute toxicity tests reported by Dacre (1980).

Aquatic toxicity assessments of munitions wastewater components conducted by Liu et al. (1983) also determined a 96-hour LC50 for fathead minnows to be 3 mg/L. The 48-hour EC50 value for Daphnia magna was reported to be 11.9 mg/L.

Pedersen (1971) reported a range of median tolerance limits (TLMs) for bluegills from 2.3 to 2.7 mg/L, depending primarily on water temperature.

No data were found regarding the effects on aquatic organisms of chronic exposure to 2,4,6-TNT.

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3.1.4 Dose-Response Assessment

An ADI of 1.4×10^{-3} mg/kg/day was calculated for 2,4,6-TNT. This value was based on a no-observed-adverse-effect level (NOAEL) of 1.4 mg/kg/day in a 90-day rat-feeding study (Dilley et al., 1978). A safety factor of 1,000 was applied to the NOAEL in this subchronic study to calculate the ADI for human beings (Vettorazzi, 1976). This ADI served as a basis for the Interim Environmental Criteria for 2,4,6-TNT established by U.S. Army Medical Bioengineering Research and Development Laboratory (USAMBRDL). Rosenblatt and Small (1981) discussed inclusion of an additional safety factor of 100 in calculating the ADI for 2,4,6-TNT for application in an assessment at Alabama Army Ammunition Plant (AAAP), based on microbial mutagenesis of 2,4,6-TNT observed in Ames test studies by Dilley et al. (1979). In a review of the AAAP task order from USATHAMA, EPA Region IV commented that the use of such a mutagenicity safety factor results in a value that is "overly conservative for the effects the ADI is designed to address...." Based on this EPA analysis, a mutagenicity safety factor was not applied in calculating the ADI for 2,4,6-TNT at WVOW. 2,4,6-TNT is currently undergoing further testing for mutagenicity under the National Toxicology Program (NTP), but there is at present insufficient evidence of carcinogenicity of 2,4,6-TNT to develop a cancer potency factor, and the ADI is based on noncarcinogenic effects.

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3.2 DINITROTOLUENES (DNTs)

3.2.1 Pharmacokinetics

The metabolism of DNTs in vitro suggests that both hepatic and microfloral enzymes can reduce DNT to amino toluenes through potentially toxic nitroso and hydroxylamino intermediates (Rickert et al., 1984). In a study of 2,4-DNT metabolism by Lee et al. (1978a), radioactive-labeled 2,4-DNT was concentrated in the liver and kidney following oral administration in dogs, mice, and rats. Primary transformation reactions were reduction of one or both nitro groups to amines and oxidation of the methyl group to a benzyl alcohol or benzoic acid. Secondary conjugation reactions occurred before urinary excretion. Schut et al. (1982) examined the metabolism and distribution of 2,4-DNT in mice. The distribution of 2,4-DNT in blood, liver, kidneys, lungs, adipose tissue, and small and large intestine showed no evidence for preferential uptake or retention at any dose tested. Doses of 2,4-DNT were rapidly and extensively metabolized by liver and small intestine. The blood and lungs contained much higher levels of unchanged 2,4-DNT, and, in most cases, the extent of 2,4-DNT metabolism was similar in the two tissues, suggesting that the lung is not an active site of 2,4-DNT metabolism.

In another study, Schut et al. (1983) studied the metabolism of the 2,6-DNT isomer in mice. The distribution of the compound in various tissues showed no evidence for preferential uptake or retention at any dose (maximum total dose of 6,000 mg/kg administered over a 30-week bioassay period).

Animal studies indicate significant species and strain differences in the excretion of DNT isomers. Rats have been observed to excrete in urine up to 80 percent of various DNT isomers within 24 hours and 55 to 90 percent of 2,4- and 2,6-DNT within 72 hours (Lee et al., 1975; Rickert and Long, 1981). Wistar rats studied by Mori et al. (1981), however, excreted only 13 percent of a dose of 2,4-DNT in urine and 21 percent in feces within 24 hours. Strain A mice excreted 52 to 70 percent of 2,4-DNT up

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to 100 mg/kg) in urine within 4 hours of administration (Schut et al., 1982). However, strains of mice studied by Lee et al. (1975) excreted only about 10 percent of a dose of 2,4-DNT in urine and about 80 percent in feces. In the same study, urine was found to be the major route of 2,4-DNT excretion in rabbits, dogs, and monkeys.

A variety of urinary metabolites of DNT have been identified in animal experiments. Mori et al. (1981) found the following urinary metabolites in rats administered 2,4-DNT: 2-amino-4-nitrotoluene; 4-amino-2-nitrotoluene; 2,4-diamino toluene; 2,4-dinitrobenzyl alcohol; 2-amino-4-nitrobenzyl alcohol; 4-amino-2-nitrobenzyl alcohol; 4-acetylamino-2-nitrotoluene; 4-acetylamino-2-aminotoluene; and 4-acetylamino-2-amino benzoic acid. Lee et al. (1978a) administered 2,4-DNT to mice, rats, rabbits, dogs, and monkeys and identified and quantified the urinary metabolites. The predominant metabolite in all species except mice was 2,4-dinitrobenzyl alcohol glucuronide (20 to 33 percent of dose). In mice, this metabolite accounted for only 2.6 percent of the dose. Aminonitrobenzyl alcohol glucuronides accounted for 8 to 19 percent (2.8 percent in mice) of the dose. Other metabolites recovered were the aminonitrotoluenes, 2,4-diaminotoluene, 2,4-diaminobenzyl alcohol, and 2,4-dinitrobenzoic acid.

Male and female Fischer-344 rats convert 2,4-DNT into four major metabolites and 2,6-DNT into three major metabolites (Long and Rickert, 1982). The metabolites were identified as: 2,4-dinitrobenzyl alcohol glucuronide, 2,4-dinitrobenzoic acid, 2-amino-4-nitrobenzoic acid, 4-acetylamino-2-nitrobenzoic acid, 2,6-dinitrobenzyl alcohol glucuronide, 2,6-dinitrobenzoic acid, and 2-amino-6-nitrobenzoic acid, respectively.

3.2.2 Human Health Effects

There is growing evidence from laboratory animal studies that DNTs are carcinogenic, although the potency of various isomers in cancer induction has not been well characterized. Three chronic bioassays of DNT have

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been reported, with different results. The findings of each of these studies are summarized below.

In one study [Chemical Industry Institute of Toxicology (CIIT), 1979], technical-grade DNT [tDNT consists of 76.4 percent 2,4-DNT, 18.8 percent 2,6-DNT, 2.4 percent 3,4-dinitrotoluene (3,4-DNT), 1.5 percent 2,3-dinitrotoluene (2,3-DNT), 0.7 percent 2,5-dinitrotoluene (2,5-DNT), and less than 0.1 percent 3,5-dinitrotoluene (3,5-DNT)] was found to be hepatocarcinogenic in Fischer rats. At the highest dosage (35 mg/kg/day), two male rats developed hepatocellular carcinomas as early as 26 weeks. After 1 year of feeding, all of the high-dose male rats had developed carcinomas, and 50 percent of the high-dose females had developed hepatocellular carcinomas. Nearly all the female animals without hepatocellular carcinomas had neoplastic nodules after 1 year. The mid-dose group (14 mg/kg/day) had a high incidence of hepatic neoplasms after 2 years of continuous feeding.

Another feeding study sponsored by the National Cancer Institute (NCI) used 96-percent-pure 2,4-DNT (NCI, 1978) in mice (B6C3F1) and rats (Fischer 344). Rats did not show a statistically significant increase in hepatic neoplasms when administered DNT at 0.008-percent and 0.02-percent levels for 78 weeks. Both low- and high-dose male rats did not have an increased incidence of fibromas in the skin and subcutaneous tissues. The females had increased incidence of fibroadenoma of the mammary glands. The DNT-fed mice did not have a significant increase of any neoplasms when fed DNT at dietary levels of 0.008 percent and 0.04 percent for 78 weeks.

In a third study by Midwest Research Institute (Ellis, 1979b), rats and mice were fed diets containing 15, 100, or 700 parts per million (ppm) of a mixture containing 98 percent 2,4-DNT and 2 percent 2,6-DNT for up to 2 years. Only the female rats at the high dose level showed significant increases in hepatocellular carcinomas. The middle-dose group of male mice had a statistically significant elevation in kidney tumors.

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While each of these studies demonstrated a carcinogenic effect for orally administered DNT, the site of tumor formation and the potency of the DNT varied among studies. Rickert et al. (1984) state that these studies suggest that an isomer other than 2,4-DNT is responsible for hepatocarcinogenicity of tDNT.

CIIT recently conducted a series of 1-year rat-feeding studies using tDNT, 2,4-DNT (99.9-percent pure), and 2,6-DNT (99.9-percent pure) (Popp and Leonard, 1983). The 2,6-DNT (14 or 7 mg/kg/day) was found to be hepatocarcinogenic, but 2,4-DNT (27 mg/kg/day) was not. Rickert et al. (1984) conclude that the hepatocarcinogenicity of tDNT is predominately due to the 2,6-DNT isomer. They hypothesize that the 2,6-DNT-initiated sites in the CIIT bioassay may have progressed to neoplasms somewhat more rapidly due to the combined promoting activity of 2,6- and 2,4-DNT. In addition, the absence of hepatocarcinogenicity of 2,4-DNT in the NCI bioassay may have been due to the lack of initiated sites.

A comprehensive study of the mutagenicity of dinitrotoluenes conducted by Couch et al. (1981) demonstrated that all isomers of DNT and tDNT are weakly mutagenic in at least one of the base pair substitutions or frame shift strains in the Salmonella/microsome plate incorporation assay. DNT isomers exhibited no activity in various mammalian cell culture genotoxicity assays (Couch and Abernathy, 1982; Lee et al., 1978a; Lee et al., 1978b; Bermudez et al., 1980).

The DNTs exert effects on the hematologic system, including methemoglobinemia with eventual cyanosis. Lee et al. (1978a) investigated the effects of orally administered 2,4-DNT in dogs, rats, and mice. The main target organs were the erythrocytes (methemoglobinemic leading to anemia), the testes (depressed spermatogenesis), and the neuromuscular system. The highest no-effect levels were 5 mg/kg/day in male and female dogs, less than 34 mg/kg/day in male rats, less than 38 mg/kg/day in female rats, 137 mg/kg/day in male mice, and 147 mg/kg/day in female mice.

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Galuzova (1963) fed rabbits 0.5 mg/kg and 0.05 mg/kg DNT (composition and purity unknown) daily for 8 months. The rabbits gained weight at the normal rate, but a study of the organs showed well-defined protein and fatty dystrophy of the liver cells.

A low-grade regenerative anemia has been observed in rats fed 2,4-DNT at constant dietary levels of 0.07 percent (Ellis et al., 1979b) or tDNT at constant daily intakes of 35 mg/kg/day (NCI, 1978). Similar findings were obtained when dogs were fed 10 mg/kg of 2,4-DNT daily for 3 months (Ellis et al., 1979b).

Neuromuscular effects of 2,4-DNT were found in dogs (Ellis et al., 1979a and 1979b), with the first symptoms appearing at a dose level of 10 mg/kg/day of 2,4-DNT administered for 8 weeks. In most cases, toxic signs regressed within days or a week after dosage.

The signs of acute DNT intoxication include central nervous system depression, respiratory depression, and ataxia. The lethal doses of 2,4- and 2,6-DNT reported by Ellis et al. (1979a) are presented in the table below.

Acute Toxicity of DNT Isomers

Species	Isomer	LD50 (mg/kg)	
		Male	Female
Rats	2,4-DNT	568	650
	2,6-DNT	535	795
Mice	2,4-DNT	1,954	1,340
	2,6-DNT	621	807

Rickert et al. (1983) compared macromolecular covalent binding of 2,6-DNT with that of 2,4-DNT after oral administration to male rats. Covalent binding to hepatic ribonucleic acid (RNA), deoxyribonucleic acid (DNA), and

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and protein was proportional to dose for each isomer, but the binding due to 2,6-DNT was always 2 to 5 times higher than that due to 2,4-DNT. There was no selectivity of either isomer for any macromolecules. The data suggest that covalent binding to hepatic macromolecules qualitatively reflects the differences in genotoxicities between the two isomers (2,6-DNT being highly genotoxic and 2,4-DNT less genotoxic).

Both the 2,4- and 2,6-DNT isomers have been observed to affect reproductive systems in experimental animals. Degeneration of the testes and depressed spermatogenesis were found in dogs administered 2,6-DNT at 100 mg/kg/day for 13 weeks (Lee et al. 1976). In the same study, rats and mice were also observed to have decreased spermatogenesis when fed 2,6-DNT in concentrations of 155 mg/kg/day and 289 mg/kg/day, respectively.

Dogs administered up to 5 mg/kg/day of 2,4-DNT for 13 weeks also had a decrease in spermatogenesis (Lee et al., 1978a). Rats fed up to 265 mg/kg/day of 2,4-DNT for 13 weeks had severe atrophy of the testes with no germ cells in any stage of spermatogenesis. Mice showed no testicular lesions at DNT dosages of 468 mg/kg/day for 13 weeks.

In a study by Ellis et al. (1979a), no testicular lesions were noted in dogs administered up to 10 mg/kg/day 2,4-DNT for 2 years. Rats administered up to 45 mg/kg/day of 2,4-DNT for 2 years had decreased spermatogenesis. Mice fed up to 900 mg/kg/day for 2 years had atrophy of the testes and nonfunctioning follicles in females.

A comprehensive three-generation reproduction study in rats showed no effects of 2,4-DNT on fertility, liveborn index, weight at birth, litter size, weight at weaning, or sex ratio (Ellis et al., 1979b).

A dominant lethal assay in mice with tDNT, 2,4-DNT, and 3,5-DNT showed no treatment induced increases in dominant lethals, indicating no mutagenic

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effects in male germ cells (Soares and Lock, 1980). Treatment with 2,4-DNT did reduce the percent of fertile matings. The findings of this study indicate that 2,4- and 2,6-DNT are testicular toxicants but are not genetically active in male germ cells.

Epidemiological studies of men occupationally exposed to DNT failed to demonstrate effects on fertility or reproductive health (Ahrenholz and Channing, 1980; Hamill et al., 1982; and Levine et al., in press).

3.2.3 Environmental Toxicity

3.2.3.1 Toxicity to Wildlife

No information was found in the literature regarding the toxicity of dinitrotoluenes to wildlife. In the absence of relevant information, extrapolation of toxicity data from laboratory animal studies to mammalian wildlife is useful.

Laboratory animal studies indicate that, in small oral doses, dinitrotoluenes are readily metabolized and excreted with no evidence of preferential uptake or retention. Chronic feeding studies, however, showed carcinogenic effects in concentrations as low as 14 mg/kg/day for 2,4-DNT in rats (CIIT, 1979) and 7 mg/kg/day for 2,6-DNT (Popp and Leonard, 1983). The results of carcinogenicity testing on 2,4-DNT are equivocal, since most studies have been conducted using 2,4-DNT containing impurities of 2,6-DNT and other DNT isomers. The sites of tumor formation also varied among studies.

No-effect levels for orally administered 2,4-DNT have been reported as 5 mg/kg/day in dogs, less than 34 mg/kg/day in rats, and 137 mg/kg/day in mice (Lee et al., 1978a). Neuromuscular effects of 2,4-DNT have been observed in dogs fed 10 mg/kg/day for 8 weeks (Ellis et al., 1979a and 1979b).

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Reproductive effects, primarily atrophy of the testes and decreased spermatogenesis, have been observed in experimental animals administered 2,4- and 2,6-DNT. Lee et al. (1978a) found decreased spermatogenesis in dogs fed up to 5 mg/kg/day of 2,4-DNT for 13 weeks. In another study, the same researcher noted these effects in dogs administered 2,6-DNT at 100 mg/kg/day for 13 weeks (Lee et al., 1976). Rats and mice appear to be less sensitive to both isomers.

3.2.3.2 Aquatic Toxicity

Aquatic toxicity data are limited for the DNTs. Ambient water quality criteria for protection of aquatic life have been developed by EPA (1980a), based primarily on two studies (U.S. Army Research and Development Command, 1976; EPA, 1978).

The U.S. Army Research and Development Command (1976) reported a 48-hour EC50 value for Daphnia magna for 2,4-DNT as 35,000 µg/L. The 96-hour LC50 for the fathead minnow exposed to 2,4-DNT is 31,000 µg/L. Based on EPA data (1978), 2,3-DNT appears to be up to two orders of magnitude more acutely toxic to freshwater fish and invertebrates than 2,4-DNT.

Liu et al. (1983) evaluated the acute and chronic freshwater aquatic toxicity for a number of munitions compounds, including 2,4-DNT and 2,6-DNT. The 48-hour EC50 values for Daphnia magna were 47,500 µg/L for 2,4-DNT and 21,800 µg/L for 2,6-DNT. The 96-hour LC50s for fathead minnows were 32,800 µg/L for 2,4-DNT and 18,500 µg/L for 2,6-DNT.

The only chronic toxicity value identified (230 µg/L) was reported for 2,3-DNT in an embryo-larval test with the fathead minnow (EPA, 1978). EPA (1978) also reported that cell numbers of the freshwater alga, Selenastrum capricornutum, were reduced by 50 percent at a concentration of 2,3-DNT of 1,370 µg/L. Inhibition of chlorophyll-a occurred at a concentration of 1,620 µg/L.

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Liu et al. (1983) conducted bioconcentration tests for 2,4-DNT in several freshwater aquatic species. Bioconcentration factors (BCFs) obtained with bluegills and D. magna were 58 and 13, respectively. The BCF for S. capricornutum was much higher, at 2,507. The researchers speculated that this result may be attributable to sorption of 2,4-DNT to the outer cell walls.

3.2.4 Dose-Response Assessment

The acceptable dose for 2,4-DNT is based on data used to develop the EPA ambient water quality criterion (EPA, 1980a). The water quality criterion for protection of human health is 0.11 µg/L, or 1.1×10^{-4} mg/L at the 10^{-6} risk level. The acceptable dose at the 10^{-6} cancer risk level is calculated by dividing the risk level by the carcinogenic potency factor, q^* , published in the criteria document. It was determined by recalculation of the criterion that the published potency factor of 3.6965×10^{-2} (mg/kg/day) $^{-1}$ published in the EPA Ambient Water Quality Criteria document for DNT was erroneous. The actual value, determined by the EPA Carcinogen Assessment Group, is 0.31 (mg/kg/day) $^{-1}$ (EPA, 1986). The resulting acceptable dose for 2,4-DNT at the 10^{-6} cancer risk level is 3.2×10^{-6} mg/kg/day.

The water quality criterion for 2,4-DNT was based on a bioassay using 98-percent-pure 2,4-DNT (Lee et al., 1978b) (the water quality criteria document erroneously states that practical-grade DNT was used). In this study, hepatocellular carcinoma, fibromas, and fibroadenomas were observed in rats. The water quality criteria document states: "The influence of the other isomers and impurities on the carcinogenic activity of technical-grade 2,4-DNT cannot be properly assessed at this time."

In another rat-feeding study by NCI (1978), 2,4-DNT (96-percent purity) was again found to be carcinogenic, resulting in an increased incidence of fibromas in the skin and subcutaneous tissues in male rats.

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fibroadenomas of the mammary gland in female rats. Although these tumors were benign, the Data Evaluation/Risk Assessment subgroup of the Clearinghouse on Environmental Carcinogens commented that the tumors must be viewed with concern, especially since the maximum tolerated dose may not have been attained and because of the potential for considerable human exposure to 2,4-DNT (NCI, 1978).

A third study conducted by CIIT (1979) demonstrated the hepatocarcinogenicity of tDNT (approximately 76-percent 2,4-DNT; 19-percent 2,6-DNT; and 5-percent other isomers). In a more recent series of 1-year feeding studies using tDNT, 2,4-DNT (99.9-percent purity), and 2,6-DNT (99.9-percent purity), CIIT concluded that the observed hepatocarcinogenicity is primarily due to the 2,6-DNT isomer. The details of this study have not yet been published, but a summary of results has been published in abstract form (Popp and Leonard, 1983). The study does not appear to refute evidence that 2,4-DNT is carcinogenic, but it demonstrates that the sites of cancer formation and the cancer potency may be different for 2,4-DNT and 2,6-DNT. Thus, for the purposes of this assessment, it is assumed that the data used as a basis for the 2,4-DNT water quality criterion is the best available information. Furthermore, the potency factor calculated on the basis of these data may represent an overly conservative estimate of carcinogenic risk since a portion of the observed carcinogenic response may, in fact, be due to impurities of 2,6-DNT in the 2,4-DNT administered in the feeding studies by Lee et al. (1978b).

Few toxicological data are available for pure 2,6-DNT. The most informative of the available studies was performed by Popp and Leonard (1983). In this 1-year rat-feeding study, 99.9-percent-pure 2,6-DNT was found to be hepatocarcinogenic at doses as low as 7 mg/kg/day.

Since the available data pertaining to 2,6-DNT are so limited at this time, calculation of an acceptable dosage is extremely difficult. It is

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critical that the value be protective against the suspected carcinogenic effects of the 2,6-DNT isomer. Two approaches were considered by ESE for developing an acceptable dosage for this compound. One approach, and the simplest, was based on the assumption that 2,4- and 2,6-DNT have similar cancer potencies and, thus, would have similar acceptable dosages. Using this assumption, the sum of the concentrations of 2,4- and 2,6-DNT in water must be less than the established EPA water quality criterion for 2,4-DNT. There are, however, no data to support this assumption of similar degrees of toxicity. In fact, the previously mentioned 1-year rat-feeding study by Popp and Leonard (1983) demonstrated that 99.9-percent-pure 2,6-DNT was carcinogenic at concentrations five times lower than for 2,4-DNT in the Lee et al. (1978b) study (7 mg/kg/day versus 35 mg/kg/day). If these results are applied to development of an acceptable dose for 2,6-DNT, it is implied that an acceptable dosage for 2,6-DNT is five times lower than the acceptable dosage for 2,4-DNT. This approach appears to be more appropriate at this time in view of the limited toxicological data pertaining to 2,6-DNT. The resulting acceptable dosage, 6.4×10^{-7} mg/kg/day at the 10^{-6} cancer risk level [obtained by dividing the acceptable dose level for 2,4-DNT (3.2×10^{-6} mg/kg/day) by a factor of 5], appears to offer a reasonable assurance of protecting public health from the carcinogenic effects of 2,6-DNT, based on limited toxicological evidence.

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3.3 1,3,5-TRINITROBENZENE (1,3,5-TNB)

3.3.1 Pharmacokinetics

No information was found pertaining to absorption, metabolism, or extraction of TNB.

3.3.2 Human Health Effects

Toxicological studies of TNB are lacking. No information was found on the potential carcinogenicity or reproductive effects of TNB. Dilley et al. (1979) reported a mutagenic response by TNB in the Ames assay; however, additional studies corroborating this finding were not located in the literature.

The most applicable study found was a rat-feeding study by Folgeman et al. (1955) in which he observed an LD50 of 505 mg/kg for TNB.

3.3.3 Environmental Toxicity

3.3.3.1 Toxicity to Wildlife

No information was found concerning toxicity of TNB to wildlife.

3.3.3.2 Aquatic Toxicity

TNB appears to be more toxic than 1,3-DNB in tests performed by van der Schalie (1983). Static acute tests resulted in LC50s for fish that were all less than 1 mg/L. The daphnids were somewhat less sensitive, and algal-fed daphnids were still less, with 48-hour EC50s in the range of 4 to 6 mg/L.

A 10-day LC50 of 0.46 mg/L was obtained for fathead minnows and 0.52 mg/L for rainbow trout.

Early life stage tests showed significant reductions in both the survival of fry and in the overall survival of fathead minnows and rainbow trout at concentrations as low as 0.12 mg/L. The no-effect/effect concentration range for rainbow trout in early life stage tests with TNB is 0.082 to 0.17 mg/L.

In chronic toxicity testing with daphnids, the no-effect/effect range is 0.47 to 0.75 mg/L. Daphnids were substantially less sensitive to TNB chronic effects after a 21-day exposure than were fathead minnows and rainbow trout exposed in early life stage tests.

As with dinitrobenzene (DNB), algae appear to be highly sensitive to the effects of TNB relative to other organisms tested. Concentrations from 1.18 to 17.3 mg/L of TNB were algicidal while lower concentrations (0.10 and 0.17 mg/L) were algistatic (renewed growth occurred after exposed cells were transferred into fresh algal assay medium).

3.3.4 Dose-Response Assessment

The toxicological data available for 1,3,5-TNB are limited to short-term toxicity studies. Rosenblatt and Small (1981) derived an ADI for 1,3,5-TNB based on a rat-feeding study by Folgeman et al. (1955). The LD50 estimated for rats is 505 mg/kg. A safety factor can be derived according to the methods of Handy and Schindler (1976). This safety factor assumes a safe limit for the continuous body concentration of a toxic substance to be $5 \times 10^{-4} \times \text{LD50}$. Handy and Schindler (1976) also assume a biological half-life of 30 days, or a disappearance rate of 2.31 percent per day. The ADI is thus calculated as follows:

$$\begin{aligned}\text{ADI} &= 2.31 \times 10^{-2} \times 5 \times 10^{-4} \times \text{LD50} \\ &= 1.155 \times 10^{-5} \times \text{LD50}\end{aligned}$$

The ADI derived by Rosenblatt and Small (1981) for 1,3,5-TNB is thus 5.8×10^{-3} mg/kg/day. This safety factor results in a more conservative ADI than use of the standard uncertainty factor of 1,000 when extrapolating from acute animal toxicity data to chronic human effects. Furthermore, although calculation of ADIs from LD50 data is not the preferred method, there are no more adequate toxicological data available. The approach for estimating the ADI is the same used by Rosenblatt and Small (1981), and the ADI value is the same as the one adopted by those authors. Observed mutagenicity reported by Dilley et al. (1979) for this compound was not considered in development of this ADI, for the same reasons discussed for TNT.



3.4 1,3-DINITROBENZENE (1,3-DNB)

3.4.1 Pharmacokinetics

There is very little information available regarding the absorption and metabolism of 1,3-DNB. The isomers are readily absorbed through the skin (Hamblin, 1963), and all isomers are methemoglobin-formers (Watanabe et al., 1976). 1,3-DNB is metabolized by enzymic reduction of one or both of the nitro groups and addition of a hydroxyl group adjacent to one of the amino groups. This is followed by conjugation, and the metabolites are excreted in the urine. Parke (1961) demonstrated that 65 to 93 percent of 1,3-DNB fed to rabbits was excreted in urine, primarily as metabolites m-nitroaniline, m-phenylenediamine, 2-amino-4-nitrophenol, and 2,4-diaminophenol during the first 2 days. Approximately 1 to 5 percent of the dose was excreted in the feces. Cody et al. (1981) demonstrated a significant reduction in hexobarbital-induced sleep time in female rats ingesting more than 50 mg/L of 1,3-DNB in their drinking water, suggesting induction of hexobarbital-metabolizing enzymes. Hexobarbital is deactivated by hydroxylation (Parke, 1968), which is also a step in the metabolism of 1,3-DNB.



3.4.2 Human Health Effects

The toxicity of 1,3-DNB in humans has not been well studied. No information was found regarding the potential for carcinogenicity, mutagenicity, or teratogenicity of DNBs. The vapors of DNB isomers are highly toxic (Hamblin, 1963); the industrial hygiene standard established by the American Council of Government Industrial Hygienists (ACGIH, 1984) does not distinguish between the three isomers. As previously stated, all isomers are methemoglobin-formers and cause anemia on prolonged exposure (Watanabe et al., 1976). Other symptoms in humans are reduced vision, hearing loss, and ringing ears [National Institute for Occupational Safety and Health (NIOSH), 1977]. Liver damage has not been demonstrated in humans.

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The most comprehensive study of acute and subchronic toxicity of 1,3-DNB was conducted by Cody et al. (1981). In lethal doses, rats exhibited reduction in ambulatory motion, ataxia, weakness, dyspnea, and rapid heartbeat, followed by respiratory failure, coma, and death. The LD50 value for both sexes combined was 83 mg/kg. For rats given sublethal doses of 1,3-DNB, symptoms were less severe, and recovery occurred within 1 week.

When 1,3-DNB was added to drinking water in concentrations of 50, 100 and 200 mg/L for 8 weeks, 3 of 6 male rats receiving the highest concentration died during Week 4, and another died during Week 5. One female rat died during Week 6 and another during Week 7.

Growth rate was reduced in both sexes; at 200 mg/L of 1,3-DNB in drinking water, the animals lost weight. Reductions in hematocrit and hemoglobin levels were also observed. Enlarged spleens were present in both sexes at all concentrations. Testicular atrophy was seen, but there was no effect on ovaries.

When 3, 8, or 20 mg/L of 1,3-DNB was given in drinking water for 16 weeks, no signs of acute toxicity were seen. Moderate reductions in hematocrit and hemoglobin values were found in males after 5 and 10 weeks of 20 mg/L dosage. Enlarged spleens were again observed. Testes were reduced in weight among males at the 20-mg/L level.

In another investigation of the subchronic effects of 1,3-DNB, Kiese (1949) observed hematologic changes, nervous disorders, and convulsions in dogs given subcutaneous injections of 0.1 to 6.0 mg/kg for up to 144 days.

3.4.3 Environmental Toxicity

3.4.3.1 Toxicity to Wildlife

No data were found pertaining to the toxicity of DNB to wildlife.

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3.4.3.2 Aquatic Toxicity

van der Schalie (1983) found a fairly large range of sensitivity to DNB in organisms tested. Daphnids were found to be the least sensitive, with a 48-hour EC50 of 27.4 mg/L. The 96-hour LC50 for fathead minnows was 16.8 mg/L of DNB. For rainbow trout, chronic toxicity was estimated using a 69-day early life stage test. The range from the highest no-effect concentrations to the lowest concentration causing toxicant-related effects was 0.50 to 0.97 mg/L.

Juvenile trout tested in a 30-day exposure were even more sensitive, with a no effect-effect range for mortality of 0.16 to 0.42 mg/L. The alga Selenastrum capricornutum was about as sensitive as the most sensitive fish species, with 5-day growth reduction at concentrations of 0.97 mg/L, but not at 0.26 mg/L.

3.4.4 Dose-Response Assessment

Toxicological studies for 1,3-DNB are extremely limited. One subchronic oral rat study was identified as a potential basis for an ADI. In this study by Cody et al. (1981), increased spleen weight and decreased spermatogenesis were observed at doses between 1 and 3 mg/kg. An NOAEL of 0.4 mg/kg was reported. Using this value as the basis for an ADI, with an uncertainty factor of 1,000, gives an estimated ADI of 4×10^{-4} mg/kg/day.

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3.5 LEAD

Lead was detected in only a limited number of environmental samples at WVOW, at concentrations below established environmental criteria or similar to background levels. For this reason, the health and environmental effects of lead are discussed only briefly.

3.5.1 Pharmacokinetics

The major routes of lead absorption in the body are the gastrointestinal tract and the respiratory tract. The gastrointestinal absorption of lead is considerably greater in children than in adults, and during fasting than with meals. The absorption of lead from the gastrointestinal tract is also greatly influenced by dietary levels of numerous substances, such as calcium, iron, fats, and proteins (Barltrop and Khoo, 1975). Very few studies of dermal absorption of lead in humans or animals have been conducted.

Approximately 90 to 95 percent of the total adult body burden of lead with long-term exposure is in the bones. In children, this value is only approximately 72 percent (Barry, 1975).

In blood, lead is localized in the erythrocytes. The ratio of lead concentration in the cell to that in the plasma is approximately 16:1. No other notably high affinities are recognized, although the liver and kidney have somewhat higher than average concentrations (Hammond and Beliles, 1980).

Lead crosses the placenta readily, and concentrations of lead in the newborn have been found similar to maternal blood levels (Rosen and Sorell, 1978; Cavalleri et al., 1978; EPA, 1980b).

The extraction of lead is by way of the bile and the urine and by exfoliation of epithelial tissue, including hair. In adults, urinary excretion is the dominant route of excretion. In infants, GI excretion of lead is greater than urinary excretion (Rabinowitz et al., 1978).
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3.5.2 Human Health Effects

Lead exposure in humans, as well as animals, can result in toxic effects in the brain and central nervous system, the peripheral nervous system, the kidneys, and the hematopoietic system (Hammond and Beliles, 1980). Chronic exposure to inorganic lead by ingestion or inhalation can cause lead encephalopathy, with severe cases resulting in permanent brain damage. Lead poisoning may cause peripheral neuropathy in adults and children. Low level exposures in children may cause clinically undetectable permanent learning disabilities. Epidemiological studies indicate the moderately elevated lead exposure in infants and young children [blood lead levels of 40 to 80 micrograms per 100 milliliters ($\mu\text{g}/100\text{ mL}$)] may cause diminished performance in psychometric performance and neurologic tests (Bornschein *et al.*, 1980).

Exposure to lead at high concentrations may result in progressive kidney damage and possibly kidney failure. Anemia due to inhibition of hemoglobin synthesis and a reduction in the lifespan of circulating red blood cells is an early sign of lead poisoning (Hammond and Beliles, 1980).

Several lead salts have been shown to be carcinogenic in mice and rats, causing tumors of the kidneys. Data concerning the carcinogenicity of lead in humans are inconclusive, however. There is equivocal evidence that lead exposure causes genotoxicity in humans. Evidence indicates that lead poses a reproductive hazard and exerts a toxic effect on conception, pregnancy, and the fetus (EPA, 1984).

3.5.3 Environmental Toxicity

3.5.3.1 Toxicity to Wildlife

Although lead has been found in the tissues of many wild animals, including birds, mammals, fish, and invertebrates, reports of poisoning usually involve waterfowl. The effects of lead on wildlife are not discussed in detail in this report, since environmental samples did not indicate detectable concentrations outside the Burning Grounds.

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3.5.3.2 Aquatic Toxicity

Freshwater vertebrates and invertebrates are more sensitive to lead in soft water than in hard water. At a water hardness of about 50 mg/L (as calcium carbonate (CaCO_3)), the median effect concentrations of 10 species range from 140 $\mu\text{g/L}$ to 236,000 $\mu\text{g/L}$. Chronic values for Daphnia magna and the rainbow trout are 12.3 $\mu\text{g/L}$ and 83.1 $\mu\text{g/L}$, respectively, at a hardness of 50 mg/L. Freshwater algae show an inhibition of growth at concentrations above 500 $\mu\text{g/L}$. BCFs range from 42 for brook trout to 1,700 for a snail (EPA, 1985a).

3.5.4 Dose-Response Assessment

Estimates of acceptable lead exposure levels are variable and under reevaluation by EPA. An underlying premise of the current air standard (1.5 $\mu\text{g/m}^3$) and ambient water quality standard (50 $\mu\text{g/L}$) is that children are the most sensitive subpopulation and that a blood lead level of ≤ 30 micrograms per deciliter ($\mu\text{g/dL}$) has been generally considered to provide an adequate margin of safety for protection against adverse effects in children (EPA, 1980b). This target level is currently being reevaluated by EPA, since a recent review of the data has indicated potential effects at lower blood lead levels. EPA has proposed to lower the maximum contaminant level for lead in drinking water from 50 $\mu\text{g/L}$ to 20 $\mu\text{g/L}$ (EPA, 1985b).

A number of studies are being performed, or have been performed, to define the dose-response relationship between lead intake and neurological development, behavior, and learning ability. At present, there is insufficient evidence to characterize this relationship.

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4.0 ENVIRONMENTAL FATE AND TRANSPORT

In this section the environmental fate and contaminant transport for each area of concern for the WVOW RI/FS are addressed on an area-by-area basis. For each of these, the source area is characterized followed by the major environmental phenomena affecting contaminant migration from the source area and contaminant concentrations in affected media.

In this introductory section, regional geology of the site and climatic and hydrologic conditions that apply generically to all of the areas of concern are presented.

The WVOW site is located in Mason County, along the western boundary of the state adjacent to the Ohio River. Most of the site is about 1 to 2.5 mi east of the present-day river and is situated on alluvial terrace deposits; the western section lies within the floodplain of the Ohio River. The upland areas in the eastern section of the post are characterized by ridges of narrow to medium width. These ridges are remnants of an old plateau, and their slopes vary due to differential weathering of sandstones and shales. Runoff is fairly high and rapid in the dissected ridge areas. Site elevations range from about 560 feet above mean sea level (ft-MSL) along the Ohio River to about 880 ft-MSL along the northern boundary. The Ohio River has a pool elevation of 538 ft-MSL near WVOW [U.S. Geological Survey (USGS), 1968; 1975 Photorevised]. Bedrock in the area is 60 to 135 ft below land surface and is covered in consolidated river terrace alluvium. The river terraces along the Ohio River were deposited as glacial outwash and have been reworked by recent river migration and floodplain development.

The area experiences a continental climate, which is generally mild and humid. Winters are relatively mild, with intermittent snowfall.

Average wind direction is from the northwest at an average speed of 6.3 miles per hour (mph). Annual precipitation for the McClintic Wildlife

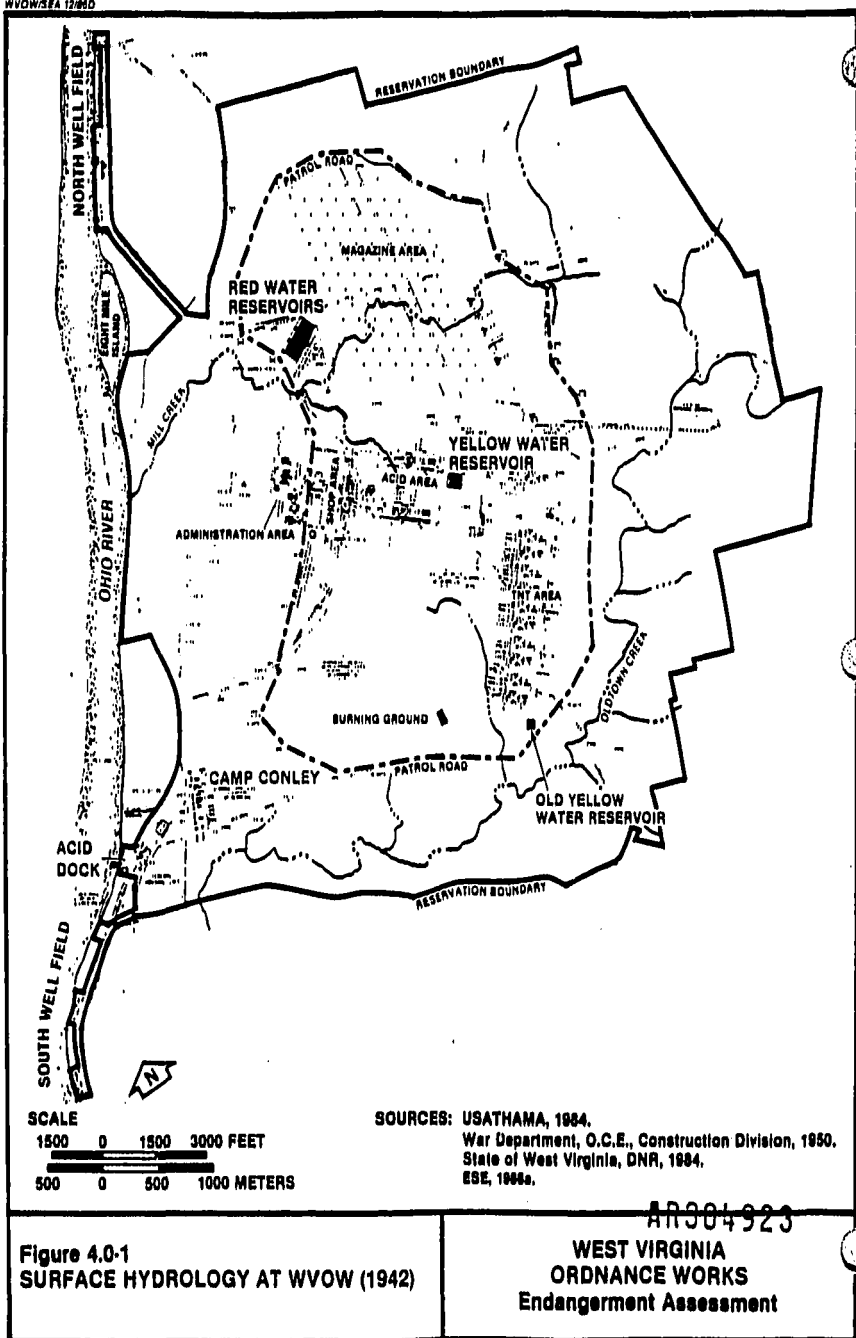
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Station is 48 inches per year (in/yr), and annual lake evaporation is 34 in/yr (annual net precipitation = 14 in/yr); which indicates a high potential for surface runoff and/or infiltration, depending on local soil characteristics (Chow, 1964).

During the period of operation in the 1940s, WVOW was drained by two major streams, their tributaries, and a number of intermittent streams (see Fig. 4.0-1). The northern half of the installation, including the magazine area and the acid area, were drained by Mill Creek and a small, unnamed tributary. Mill Creek is a tributary to the Ohio River and enters the river along the western boundary of the installation. The Ohio River is located adjacent to the installation, along the western boundary near the administration area and along the north and south well fields. As shown in Fig. 4.0-1, the southern and eastern sections of the installation were drained by Oldtown Creek. This stream and a number of smaller tributaries drained the TNT Manufacturing Area. Oldtown Creek is also a tributary to the Ohio River and intersects the river south of the installation. Three manmade surface features were also present during operations in the early 1940s. Three surface impoundments called the Red Water Reservoirs were located in the northwest section of the installation. These reservoirs had a total capacity of 30 million gallons (MG). A small reservoir called the Yellow Water Reservoir was located adjacent to the Acids Area. This reservoir had a capacity of 5 MG. A smaller water recovery reservoir, termed the "old yellow water reservoir," was located in the TNT production area; the capacity of this reservoir is unknown.

A number of manmade surface water features were constructed subsequent to installation closure in 1945. Thirty-nine ponds are currently located at the McClintic Wildlife Station (see Fig. 4.0-2). Most of these ponds were emplaced between 1953 and 1975 by the construction of impoundments and water control structures (e.g., dams and weirs) along the various drainageways. Additional pond/wetland systems have been similarly

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developed to total 39 systems as of 1985. The ponds were constructed to provide wetland habitats for various wildlife species. Currently, two of the three Red Water Reservoirs contain standing water; the northernmost reservoir is empty and has revegetated. The Yellow Water Reservoir that was present in 1945 was removed shortly after installation closure in the mid-1940s. The small water recovery reservoir located in the TNT Manufacturing Area was removed prior to 1975. Natural drainage by Mill Creek and Oldtown Creek has remained similar to the 1940s drainage, except for alteration of a number of tributaries due to pond construction.

The STORET data base (ESE, 1984a) contains instantaneous flow measurements for the Ohio River at approximately river mile 261, which is downstream from the Kyger Creek power plant at Addison, OH. The following flow data were recorded at this location, which is 2 river miles upstream of the confluence of Oldtown Creek.

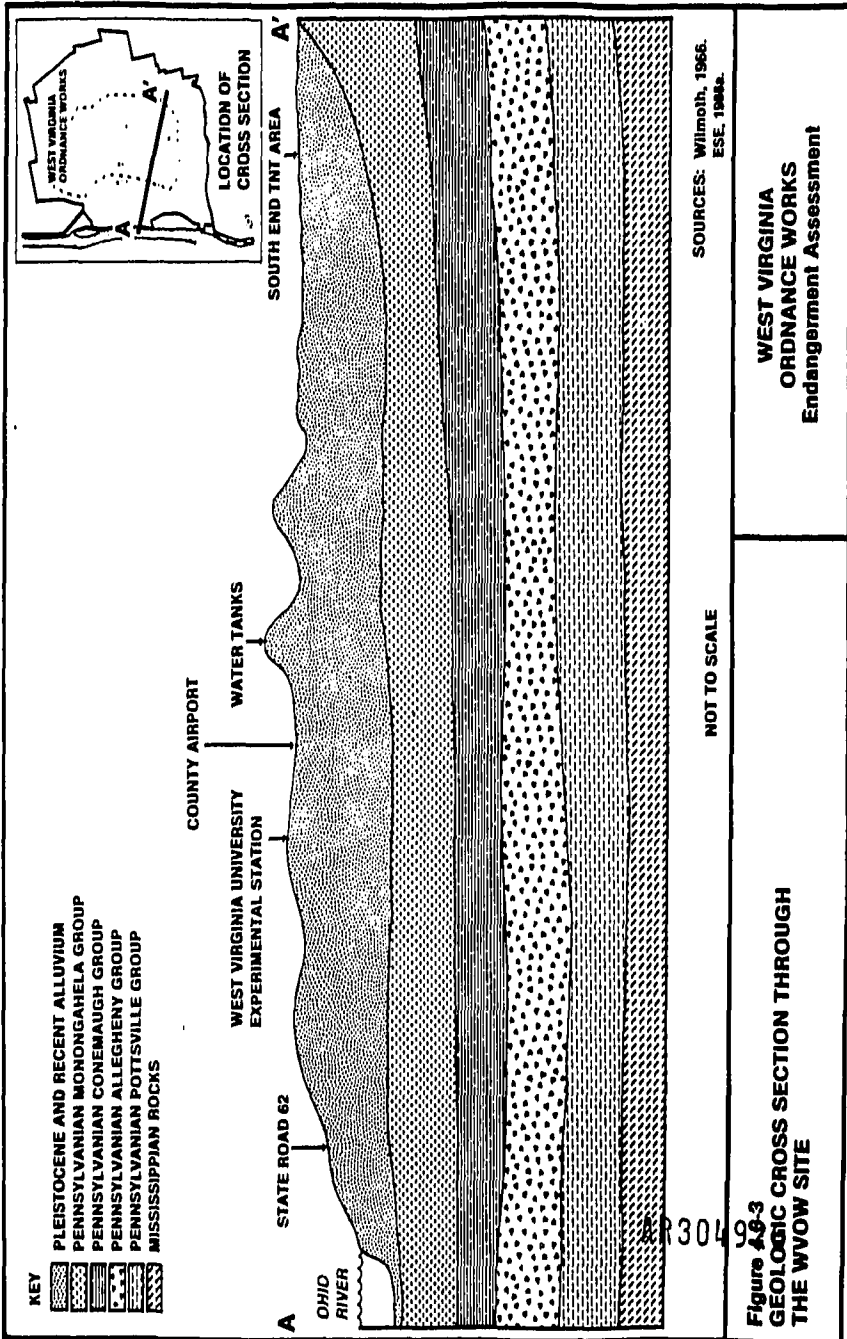
Period of Record	Number of Observations	Flow (cfs)*		
		Mean	Minimum	Maximum
June 1962-June 1964	139	82,494	12,000	328,000
July 1976-September 1983	130	59,185	3,350	244,400

*cfs = cubic feet per second.

Instantaneous flow data are available for Oldtown Creek at County Rd. 13, just southeast of the TNT Manufacturing Area. Six measurements were performed during the period April 1979 to September 1981. Mean flow was 19.8 cfs. Maximum and minimum flows measured were 57 and 0.11 cfs, respectively.

WVOW is located in the Ohio River basin, which consists of Pennsylvanian-age rocks overlain by Quaternary alluvium. The oldest exposed rocks are Pennsylvanian in age and crop out along stream valleys. Fig. 4.0-3 shows a generalized geologic cross section of WVOW developed by Wilmoth (1966). The Pennsylvanian System includes the Pottsville, Allegheny, Conemaugh,

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and Monongahela Groups. These units have a combined thickness of between 260 and 955 ft and were deposited in a freshwater environment; all the groups contain carbonaceous deposits. The Pottsville Group is the basal unit of the Pennsylvanian System. The unit consists of coarse-grained sandstone, with thin beds of coal, shale, and clay. Fig. 4.0-4 shows the major geologic units that comprise the Allegheny. The formation is between 185 and 250 ft thick in the vicinity of WVOW. The Allegheny does not crop out in Mason County and is encountered only in subsurface borings. The Conemaugh Group overlies the Allegheny Group and has a thickness of between 480 and 600 ft. The group consists of alternating sandstones, shales, and limestones, with some coal and clay units. The youngest Pennsylvanian unit is the Monongahela Group; this group also contains alternating shales, sandstones, and coal. The cross section (see Fig. 4.0-3) from Wilmoth's (1966) ground water study shows the bedrock to be part of the Conemaugh Group; however, the geologic map from the same study indicates that rocks of the Monongahela Group underly the area. Rocks from both of these groups are primarily clastic with minor amounts of limestone and coal. The Conemaugh Group contains a larger percentage of sandstone than the Monongahela Group, and both groups contain siltstones and shales. Thickness ranges from 230 to 320 ft; these units form the upland areas on the east side of WVOW.

Overlying the Paleozoic rocks at WVOW is an alluvial unit that reaches thicknesses of up to 185 ft. The alluvium is found as river floodplain deposits and elevated terraces along the Ohio River. The terraces along the Ohio River were deposited as glacial outwash to the south of the Wisconsin continental ice sheet. The alluvial deposits overlying bedrock to the east and northeast of WVOW were deposited in the channel of a pre-glacial river that flowed southward from Ohio through northern Mason County and then westward back into Ohio. The alluvium consists of a basal gravel-sand unit and decreases in coarseness from bottom to top, with a clay and silt floodplain near land surface. Fig. 4.0-5 shows a generalized cross section of these upper geologic units.

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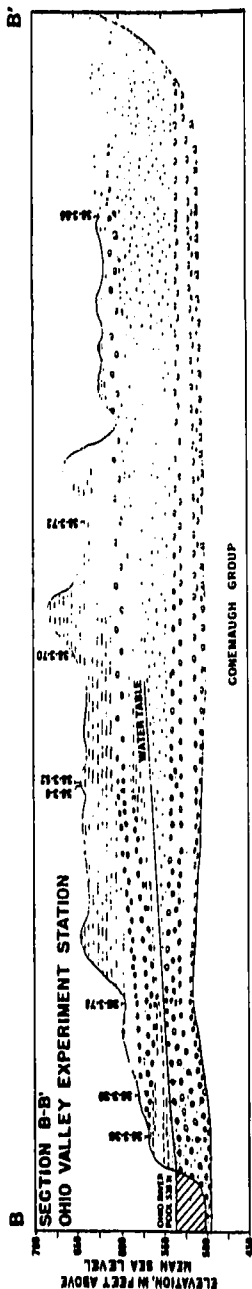
ERA	SYSTEM	SERIES	GROUP	GEOLOGIC UNIT	APPROXIMATE THICKNESS (FT)	WATER-BEARING CHARACTERISTICS
PALEOZOIC	CARBONIFEROUS	PULASKI	PULASKI	Allevium, clay, silt, sand, gravel	0 to 185	Unconfined and semiconfined ground water; principal aquifer in Ohio Valley; average well yield 200 to 300 gpm; water quality good
				Waynesboro shale, coal Clifty sandstone, shale Uniontown sandstone, shale Redstone coal Upper Pittsburgh sandstone, shale Pittsburgh coal	230 to 320	Some porous members act as aquifers; low well yield of about 10 gpm; water quality generally good, variable
				Lower Pittsburgh sandstone Pittsburgh limestone Little Pittsburgh coal, clay Connellsville sandstone Clarkburg shale, sandstone Horseman sandstone, shale Elk Lick shale, limestone Grafton sandstone Birmingham shale Upper Anna limestone Lower Anna limestone Sellersburg sandstone, shale Pine Creek shale Buffalo sandstone, shale Brush Creek coal Warren sandstone	450 to 575	Porous members serve as aquifers; low well yield of about 10 gpm; water quality variable; saline water in lower units
				Upper Freeport coal Lower Freeport shale, coal Upper Kittanning shale Upper East Lynn sandstone Middle Kittanning clay East Lynn sandstone Lower Kittanning coal	185 to 255	This yields saline water for industrial uses; low well yield of between 1 to 50 gpm
				Hamwood sandstone Stonewall-Lewiston shale, coal Coalburg coal, clay Winfield coal Cedar Grove shale Sandstone (unnamed) Shale (unnamed)	240 to 955	This yields saline brine for industrial uses; well capacity about 7 to 50 gpm
				MUCH CREEK GROUP, shale	0 to 80	Yields saline brine
				CHERRYBURN GROUP, limestone	100 to 125	Yields saline brine
				MACOMBY FORMATION, shale	0 to 50	No yield
				POCONO FORMATION, sandstone	500 to 585	Yields saline brine
				PRE-MISSISSIPPIAN ROCK		

gpm = gallons per minute.

SOURCES: Wilmoth, 1966.
ESR, 1966c.

Figure 4.0-4
STRATIGRAPHY OF CONSOLIDATED AND UNCONSOLIDATED DEPOSITS AT THE WVOW SITE

WEST VIRGINIA 304928
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SCALE

600 0 600 1,200 FEET

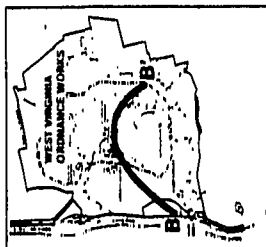
200 0 200 400 METERS

KEY

CLAY AND SILT

SAND

SAND AND GRAVEL



LOCATION OF CROSS SECTION

**SOURCES: Wilmoth, 1966.
ESE, 1966a.**

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**Figure 40-5
GEOLOGIC CROSS SECTION OF THE
UPPER GEOLOGIC UNITS THROUGH
THE WYOW SITE**

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During the phases of deposition, the Ohio River Valley probably was filled with at least 135 ft of sediment; complete sections of these deposits are preserved within the WVOW area (Wilmoth, 1966). As the river valley filled with coarse sediment, water probably became ponded and allowed finer sand, silt, and clay to be deposited.

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4.1 ACIDS AREA/YELLOW WATER RESERVOIR

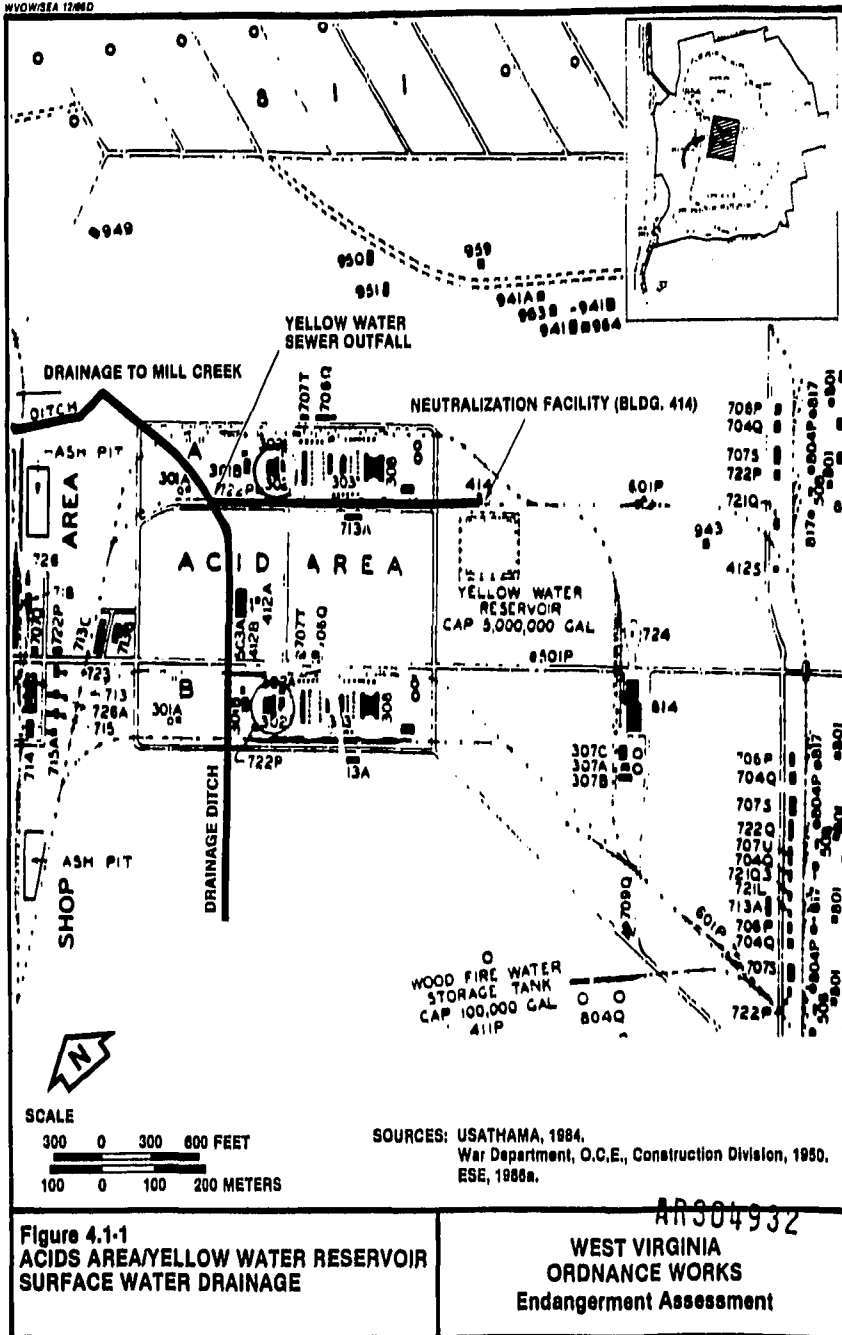
4.1.1 Site Characteristics Affecting Fate/Transport

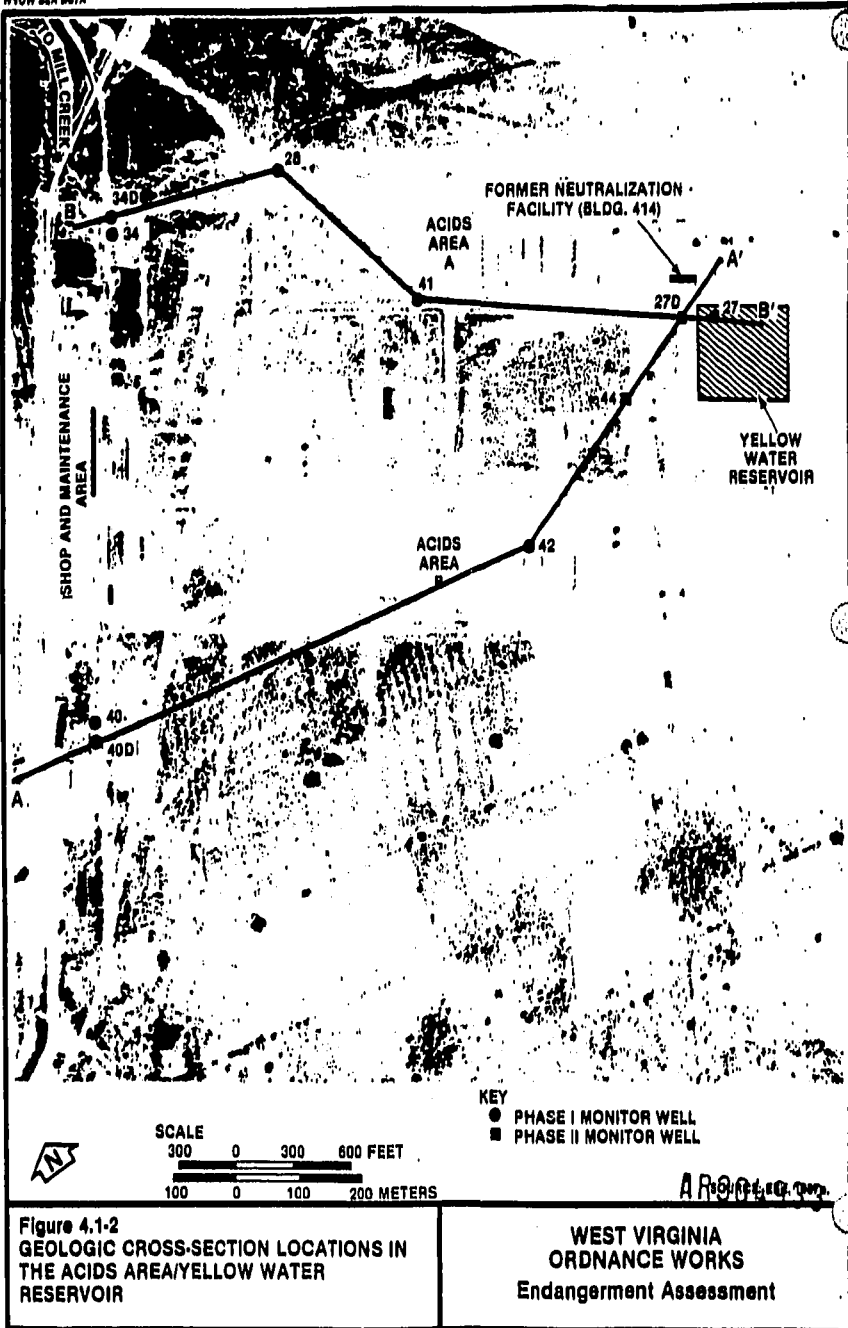
The Acids Area/Yellow Water Reservoir area is a relatively broad, flat expanse with very low relief. Ground surface elevation across the North and South Acids areas typically varies by less than 5 ft. Soils in the area consist of the Chilo Silt Loam and the Ginat Silt Loam. Both soils are classified as poorly drained, with permeabilities from slow to very slow [U.S. Soil Conservation Service (USSCS), 1961]. Surface drainage in the area is through a drainage ditch installed during WVOW construction, which conveys surface runoff to Mill Creek (see Fig. 4.1-1).

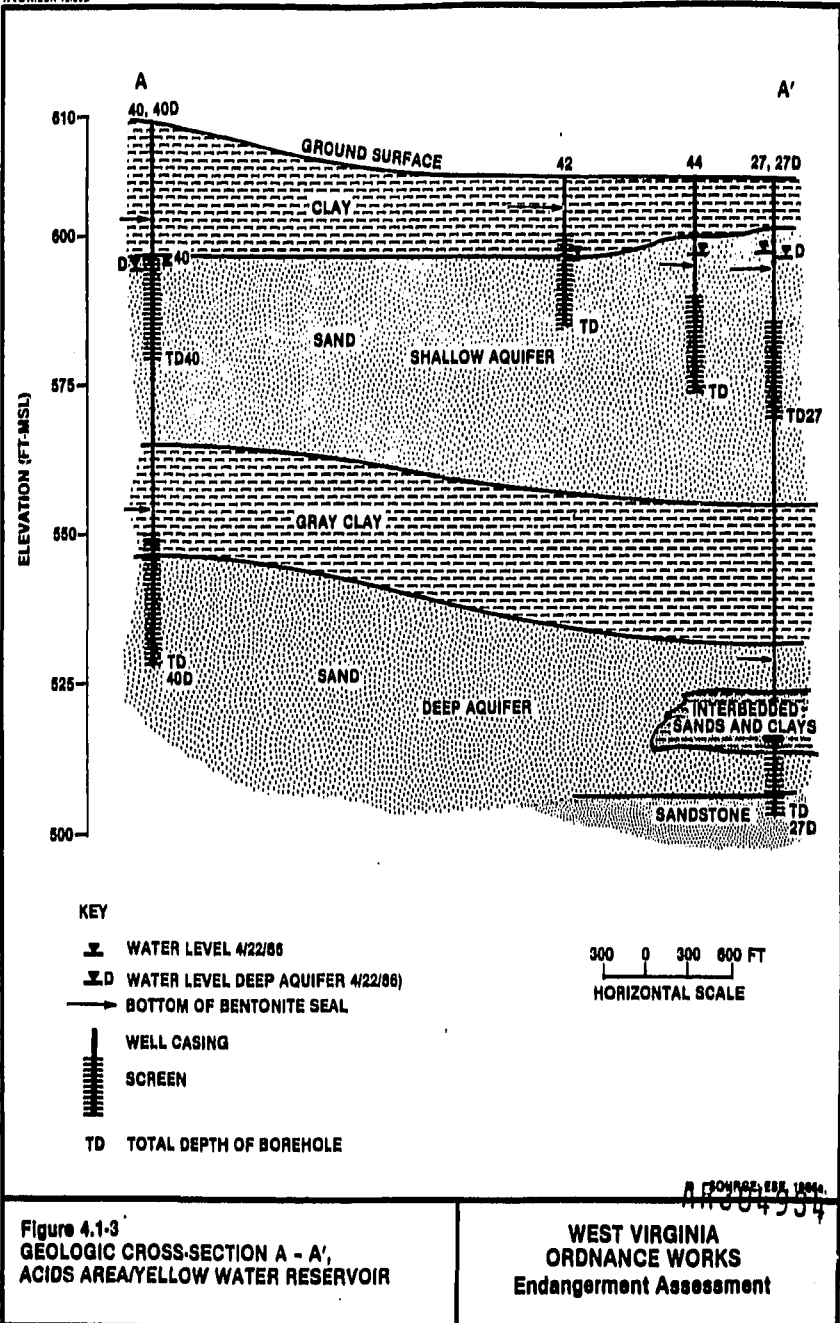
Based on the information obtained in the previous RI surveys, two geologic cross sections were prepared. As shown in the geologic cross-section index map for the Acids Area/Yellow Water Reservoir (Fig. 4.1-2), cross-section A-A' extends approximately west-east and cross-section B-B' extends approximately northwest-southeast. These cross sections are shown in Fig. 4.1-3 and Fig. 4.1-4, respectively.

GW44 is completed in the shallow aquifer. The physical characteristics of the shallow aquifer at GW44 are similar to those encountered at the remaining shallow monitor wells. These shallow sand aquifers are uniform in texture and gradation throughout this area and represent a continuous unconfined, or possibly semiconfined, aquifer.

Below the shallow sand aquifer at GW27D, a stiff gray clay, similar in textural and physical characteristics to the gray clay confining layer seen elsewhere at WVOW, was encountered at elevation 556 ft-MSL. The gray clay extends to elevation 533 ft-MSL, consistent with the thickness observed at the nearby powerhouses in Wells GW34D and GW40D. From elevations 533 to 507 ft-MSL, a deep aquifer primarily of sand with occasional interbedded clay lenses was encountered. At depth, this aquifer consists of typical glacial outwash sediments. AR004931
consisting of sandstone was encountered at elevations 507 to 506 ft-MSL. Total depth of Well GW27D was 506 ft-MSL.







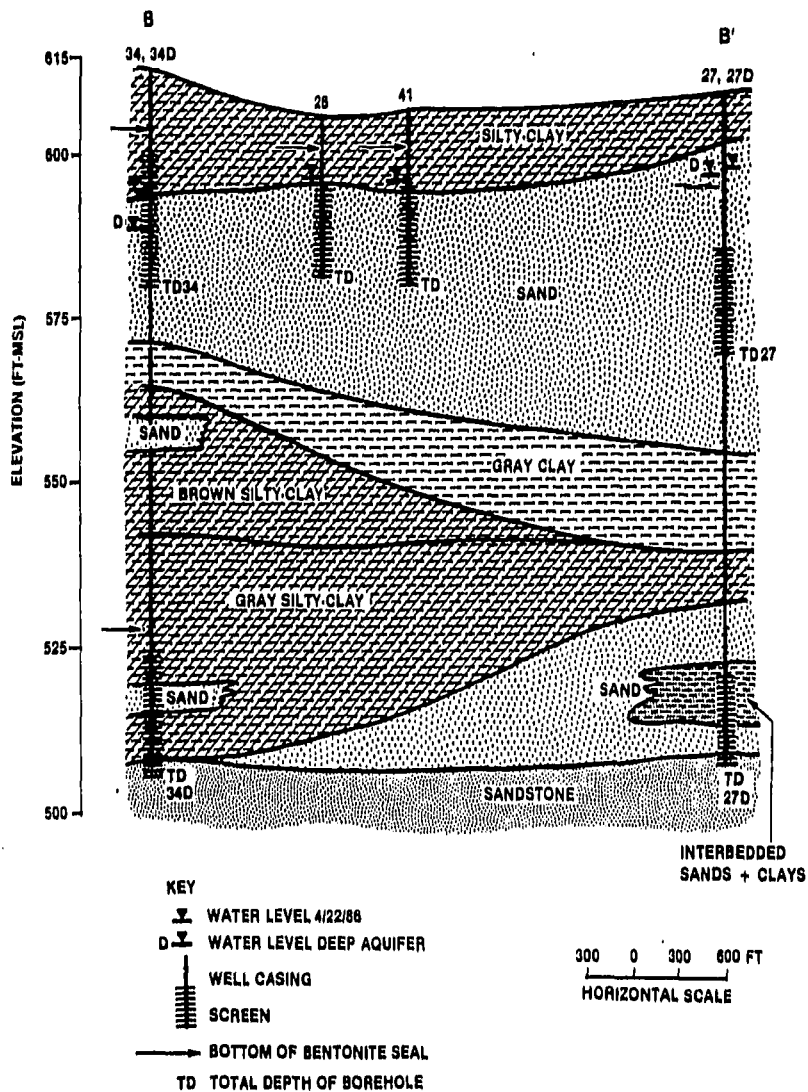


Figure 4.1-4
GEOLOGIC CROSS-SECTION B - B'
ACIDS AREA/YELLOW WATER RESERVOIR

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Endangerment Assessment

Fig. 4.1-5 represents a 2-dimensional plot showing ground water contours measured on Apr. 22, 1986. The data indicate that ground water flow is primarily to the west in the Acids Area/Yellow Water Reservoir, consistent with conditions observed in the Phase I investigation.

The shallow alluvial aquifer is separated from the deep aquifer by a gray clay confining layer. The gray clay is present at all study areas at WVOW; at the Acids Area/Yellow Water Reservoir, the clay is 23 ft thick. The gray clay acts as an effective barrier for vertical migration and should preclude any contaminant migration into the deep aquifer (ESE, 1987b).

4.1.2 Source Area Characterization

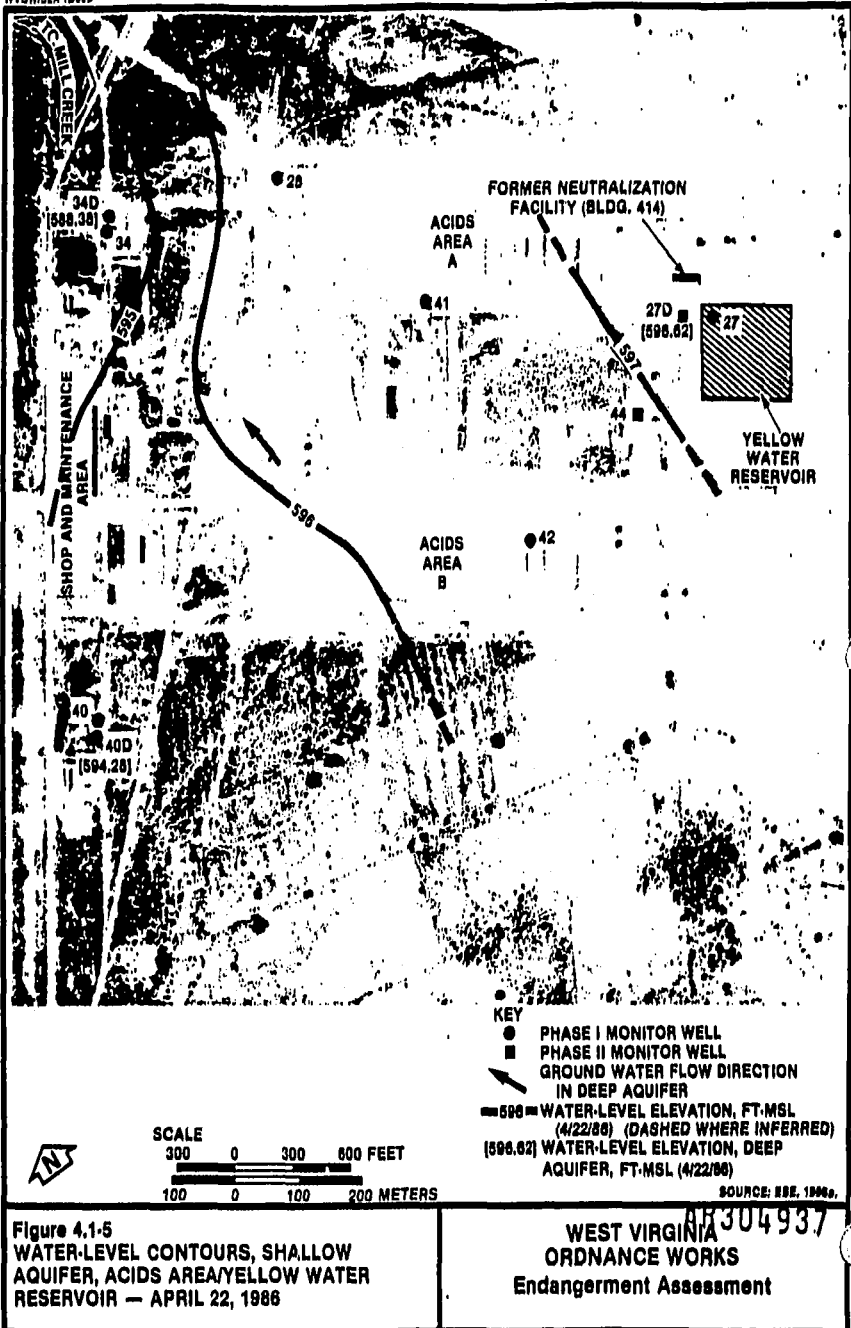
Three distinct contaminant sources have been identified in the Acids Area/Yellow Water Reservoir:

1. Contaminated surface soil adjacent to the yellow water neutralization facility;
2. The yellow water sewerline, which extends from the neutralization facility to the outfall in the Mill Creek tributary; and
3. The sediments of the Yellow Water Reservoir.

Surficial soil contamination is limited to several localized (<100 ft²) barren areas. The localized areas contain high concentrations of nitroaromatic residues (up to 1.0 percent). The contaminants are rapidly attenuated with depth; at 0.5 to 1.0 m, the contaminant concentrations are approximately one order of magnitude lower.

Soils at the invert of the yellow water sewerline near the neutralization facility are contaminated at a level of 400 µg/g (dry-weight) of total nitroaromatics. This contamination level is reduced more than an order of magnitude 1.5 m below the invert of the line. The soil below the

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sewerline is a source of contamination to the shallow aquifer, as evidenced by the nitroaromatic contamination observed in Monitor Well GW41.

Sediments at the Yellow Water Reservoir consist of 6 to 10 ft of yellow clay overlying a brown sand. With the exception of a single sample collected at the former inlet structure, the contamination was only found in the sand. Nitroaromatic contamination in the sand, based on a composite sample collected during the reactivity testing program, was present at a level of 2,830 µg/g (dry-weight) of 2,4,6-TNT. The reservoir sediments constitute a major source of the observed shallow ground water contamination.

4.1.3 Contaminant Migration Pathways

The dominant migration pathway is through infiltration of rain water, causing contaminant leaching into the shallow aquifer. The presence of multiple contaminant sources (soils, reservoir sediments, and sewerlines) minimizes the effectiveness of quantitative contaminant fate and transport predictive models. In this situation, a reliable estimate of plume extent can be based on the observed contamination at the existing monitor well network in conjunction with estimates of ground water flow rate in the shallow aquifer.

Monitor Wells GW27, GW28, GW41, GW42, and GW44 are completed in the shallow aquifer. The shallow sand aquifer is uniform in texture and gradation throughout the area and represents a continuous unconfined, or possibly semiconfined, aquifer. Highest contaminant levels (60.9 µg/L of total nitroaromatics) are present in GW27 located adjacent to the Yellow Water Reservoir. Well GW41 contained 13.9 µg/L of total nitroaromatics. The contaminant distribution in the shallow ground water aquifer is consistent with geographic and hydrologic position of each well with respect to the contamination sources. Well GW27 is located near the Yellow Water Reservoir and near the neutralization facility and

sewerline, whereas GW41 is situated adjacent to the sewerline, substantially downgradient of the remaining contaminant sources. Well GW44, located approximately 700 ft downgradient from GW27, contained no detectable nitroaromatics.

Ground water flow rates for the shallow aquifer have been estimated at 0.1 ft/day in the previous RI surveys (ESE, 1986d and 1987b). This rate of flow would result in a worst-case contaminant front located approximately 1,500 ft downgradient from the contaminant source, or twice the distance to GW44, an uncontaminated well. This variation can be attributed to retardation of the contaminants in the subsurface and to the uncertainty inherent in ground water flow rate determinations.

Although contamination of the shallow aquifer has not migrated even 700 ft downgradient of the Yellow Water Reservoir in 40 years, the apparent ground water flow direction (west-southwest, then turning westerly) indicates that shallow aquifer contamination may ultimately migrate toward the downgradient domestic well, more than 4,000 ft to the west-northwest. These wells are believed to be completed in the deep aquifer and may not be affected by contamination of the shallow aquifer system. Well construction details are not available, however, and it is possible that the quality of water drawn from these wells may be adversely affected by shallow aquifer contamination. In consideration of this unlikely, long-term risk, methods presented in Appendix A were used to estimate the ultimate steady-state concentrations in ground water in the vicinity of these wells using the following conservative assumptions:

1. Ground water migrates directly from source areas to the residential areas; actual ground water flow direction is unknown, and ground water may discharge to the drainage ditch tributary to Mill Creek.
2. Ground water contaminants would not degrade during transport. This is a very conservative assumption given an estimated half-life time of 90 to 500 years.

The method presented in Appendix A addresses dilution as a result of transverse dispersion. The effects of longitudinal and vertical dispersion are not significant to this analysis because of the steady-state assumption and long travel distances, which imply complete vertical mixing of the shallow aquifer. Nine samples of ground water have been collected from Well 27 during 1985 and 1986 (ESE 1986d and ESE 1987d). Based on the average concentrations observed and the relative toxicity of the site contaminants, 2,4-DNT is the critical ground water contaminant in the Acids Area/Yellow Water Reservoir, with an average concentration of 4.8 $\mu\text{g/L}$ in the nine samples. It will be diluted by a factor of 8 (according to procedures presented in Appendix A) during transport to the nearest domestic wells, resulting in an estimated concentration of 0.6 $\mu\text{g/L}$ in the shallow aquifer at the location of the domestic wells.

Consequently, this analysis indicates that under the no-action alternative, contaminated ground water in the shallow aquifer could migrate westerly toward residences with existing domestic wells (e.g., Wiley and Fitzgerald, see Fig. 1.2-1). The contaminant front would not reach these wells for 90 to 500 years if at all. During transport, a variety of natural mitigative processes may act to attenuate this contamination; the effectiveness of these processes cannot be quantified, with the exception of lateral dispersion. Considering this attenuative process only, it is estimated that concentrations of 2,4-DNT would not exceed 0.6 $\mu\text{g/L}$ in the shallow aquifer in the vicinity of these wells. The wells are believed to be completed in the deep aquifer, so there is little potential for contamination of the well water. As a result of this analysis and because it is based on numerous conservative assumptions, it has been concluded that these water supply wells are not threatened by contamination in the Acids Area/Yellow Water Reservoir.

AR304940

Well GW27D, screened in the deep aquifer, contained approximately 20 µg/L of total nitroaromatics in the April 1986 sampling (ESE, 1987b). Nitroaromatic compounds have not been detected in either of the deep wells at the powerhouses (GW34D and GW40D). The deep wells at the powerhouses are screened in the same aquifer as GW27D.

The observed contamination levels of GW27D when compared with GW27 indicated the possibility that contamination from the shallow aquifer may have been introduced into the deep aquifer during well installation. Subsequent time-series sampling of the well pair indicated that the contaminants in the deep aquifer were likely due to contamination introduced during well installation and did not represent a continuing source of contamination (ESE, 1987b). The gray clay confining layer constitutes an effective barrier to downward contaminant migration.

AR304941

4.2 RED WATER RESERVOIRS

4.2.1 Site Characteristics Affecting Fate/Transport

The Red Water Reservoirs area is relatively flat but slopes to the southwest toward Mill Creek, which is approximately 180 m from identified source areas. Because Red Water Reservoir sediments are the most probable source of contamination, the surface water hydrology of these manmade ponds and their interaction with the ground water system are the principal site characteristics affecting contaminant migration.

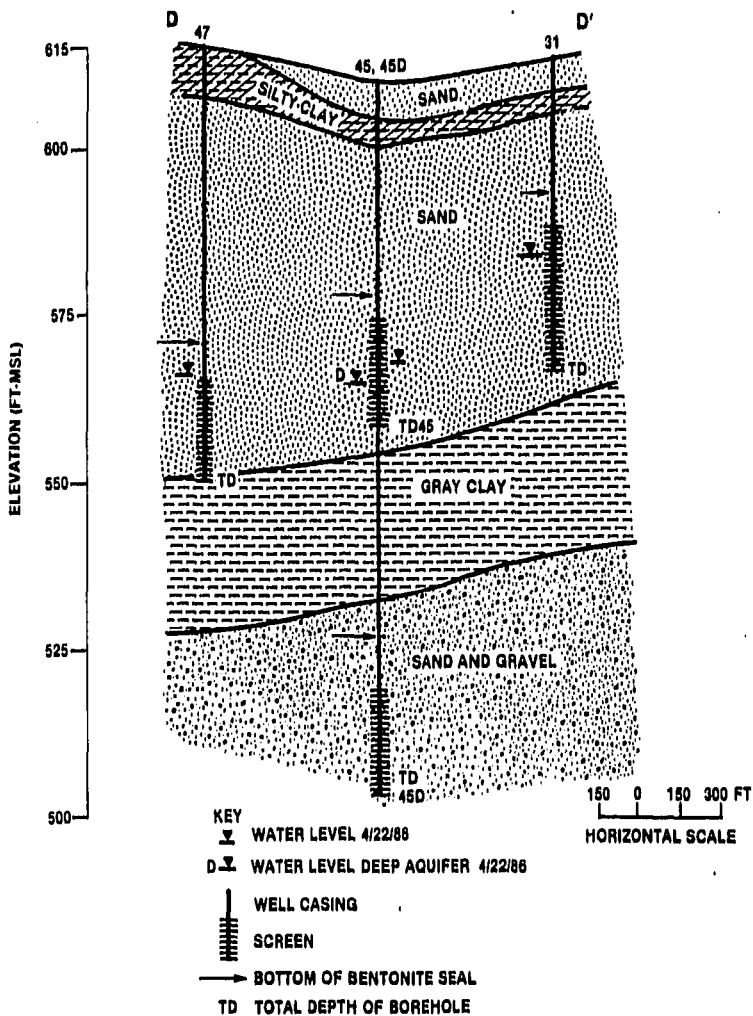
The Red Water Reservoirs are rectangular areas, bermed on all sides. Although overtopping of the berms is possible under extreme wet season conditions, there are no well-defined overflow structures, discharge channels, or evidence of erosion to indicate that such overtopping occurs. No discharge from the reservoirs was observed during site investigations. Consequently, the surface hydrologic balance of the reservoirs appears to be represented by the following relationship:

$$\text{Recharge to ground water} = \text{Precipitation} - \text{evaporation}$$

As discussed in Sec. 4.0, precipitation at WVOW is approximately 48 in/yr, and lake evaporation is 34 in/yr, resulting in 14 in/yr [0.36 meter per year (m/yr)] estimated to recharge through the reservoir sediment to the shallow aquifer. The area of the two reservoirs (Ponds 1 and 2) found to have contaminated sediments is approximately 35,000 square meters (m^2); therefore, approximately 13,000 cubic meters per year (m^3/yr) of potentially contaminated leachate is generated by the Red Water Reservoirs sediment.

A geologic cross-section trending from northwest to southeast through the Red Water Reservoirs, located between Wells GW45 and GW31, is presented in Fig. 4.2-1. This cross section approximately parallels the direction of ground water flow under the Red Water Reservoirs. The silty clay layer observed at approximately 605 ft-MSL is a low permeability layer.

AR304942



AR304943
SOURCE: EDC, 1986.

Figure 4.2-1
GEOLOGIC CROSS-SECTION D-D',
RED WATER RESERVOIRS

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that, along with the clay liner of the reservoirs, allows the Red Water Reservoirs to retain water. Immediately beneath this silty clay layer is the shallow water table aquifer. This aquifer in the Red Water Reservoirs area is approximately 15 ft thick. A deeper sand and gravel aquifer was located by boring GW45D. The deep sand and gravel aquifer, under confined conditions, is separated from the shallow water table aquifer by 23 ft of gray clay. The piezometric head in the deep sand and gravel aquifer was 3 ft lower than the water table elevation in the shallow aquifer, indicating a downward gradient of 0.06 across the confining layer and an estimated velocity of less than 0.4 foot per year (ft/yr) (see ESE, 1987b).

Ground water flow in the shallow aquifer is to the northwest, as indicated by the water-level contours observed on April 22, 1986, and August 14, 1986 (see Fig. 4.2-2 which shows the contours observed in April 1986). The velocity of ground water in the shallow aquifer was estimated by ESE (1987b) at 1.7 feet per day (ft/day). The flow of ground water under Ponds 1 and 2 can be estimated by the following equation:

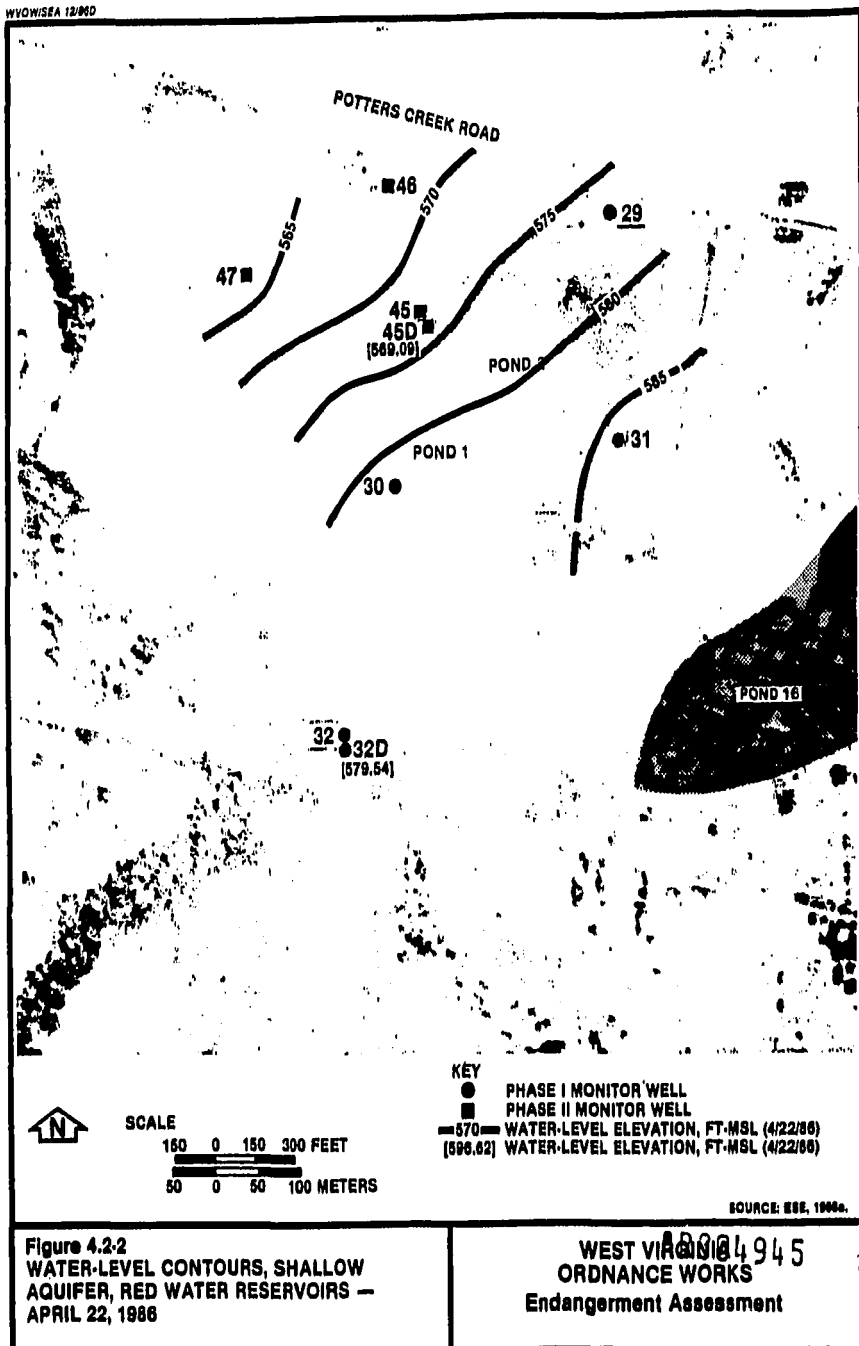
$$\text{Underflow} = W_s \times T \times K \times i$$

where: W_s = width of Ponds 1 and 2 perpendicular to the ground water flow direction (220 m, from Fig. 4.2-2);
 T = aquifer thickness (5 m, from Fig. 4.2-1);
 K = aquifer permeability (6.3 meters per day (m/day), from ESE, 1987b); and
 i = hydraulic gradient (0.02, from ESE, 1987b).

Thus:

$$\begin{aligned}\text{Underflow} &= 220 \times 5 \times 6.3 \times 0.02 \\ &= 140 \text{ cubic meters per day (m}^3\text{/day)} \\ &= 51,000 \text{ m}^3\text{/yr.}\end{aligned}$$

AR304944



Because the estimated recharge is 13,000 m³/yr, it is estimated that leachate is diluted 4:1 upon mixing with the underflow.

Ground water flow patterns northwest of Wells GW45 and GW46 are indeterminate because of the limited number of wells in this area. Furthermore, the limited information for domestic wells along State Route (SR) 62 indicates substantial variations in geology over short distances associated with bedrock highs. At the observed flow rates in the Red Water Reservoirs area, however, the ground water travel times to domestic wells along SR 62 would be approximately 4 years (refer to Sec. 4.2.3); therefore, sufficient time has passed since the Red Water Reservoirs first became contaminated for ground water contamination to have affected those wells. The lack of significant levels of contamination in the Schwartz well suggests that the contaminated flow in the shallow aquifer may discharge predominantly to Mill Creek just east of SR 62.

4.2.2 Source Area Characterization

Sources of contamination in the Red Water Reservoirs area include the sediments of Ponds 1 and 2 and soils/sediments associated with the red water sewerline. The sewerline deposits have been addressed as part of the first operable unit (ESE, 1986c) and are discussed here only to the extent that they contribute to the observed patterns of contamination.

Fifteen samples of sediments from Ponds 1 and 2 have been analyzed for nitroaromatic contamination. Of the 15 samples, 8 contained no detectable contamination. Of the seven samples with detectable nitroaromatics, none was surficial. Surficial sediments apparently have been deposited since cessation of military activities onsite and provide a clean deposit separating contaminated sediments from the water column (ESE, 1986d). The contaminated sediments are in the silty clay layer shown in Fig. 4.2-1 or the sand immediately underlying the clay deposit.

AR304946

Six of the subsurface samples contained <3.4 µg/g of nitroaromatic contamination. In these six samples, 2,4-DNT was the most frequently detected contaminant (detected in five samples), and 2,4,6-TNT was detected in three samples. No other nitroaromatics were detected. The relative composition of these sediments appears to be different from other soils/sediments at WVOW where 2,4-DNT consistently was <5 percent of total nitroaromatics. In Red Water Reservoirs subsurface sediments, 2,4-DNT may comprise 20 to 75 percent of total nitroaromatics. The average concentration of 2,4-DNT in Red Water Reservoirs sediments is 0.2 µg/g.

One Red Water Reservoirs sediment sample collected for reactivity testing exhibited a much higher level of contamination by 2,4,6-TNT at 2,210 µg/g. Other nitroaromatic compounds were not analyzed for in this sample.

4.2.3 Contaminant Migration Pathways

Contaminants are leached from the Red Water Reservoirs sediments by percolating pond water and enter the shallow water-table aquifer. The contaminants are diluted and dispersed during transport in the aquifer and migrate across the McClintic Wildlife Station boundary. Beyond that boundary, the extent of contaminant migration cannot be determined from available data. Apparently, the main part of the contaminant plume does not flow toward domestic wells serving residents along SR 62, since the concentration of nitroaromatic contaminants in the Schwartz well (refer to Sec. 1.2) in this area is much lower than would be expected from simple transport and dispersion consideration. Sufficient time has passed since the Red Water Reservoirs were used for storage of wastewater for contaminants to have migrated to the Schwartz well; therefore, the lack of significant contamination in this well indicates that the main body of the plume may discharge to Mill Creek.

AR304947

These conclusions are supported by a few rudimentary calculations pertaining to the ground water pathway, focusing on the critical contaminant, 2,4-DNT. The concentration of 2,4-DNT in Red Water Reservoirs leachate is estimated as:

$$C_1 = C_s / K_d$$

where: C_1 = concentration in leachate ($\mu\text{g/mL}$), and
 C_s = concentration adsorbed to soil solids expressed on dry-weight basis ($\mu\text{g/g}$).

From Table 2.1-3, K_d may range from 0.1 to 1.4. If $C_s = 0.2 \mu\text{g/g}$ (see Sec. 4.2.2), then $C_1 = 0.14$ to 2 mg/L . According to the water balance calculations presented in Sec. 4.2.1, leachate should be diluted 4 to 1 upon admixture with ground water in the shallow aquifer, resulting in an expected ground water 2,4-DNT concentration immediately downgradient of the Red Water Reservoirs of $35 \mu\text{g/L}$ to $500 \mu\text{g/L}$. The observed concentration at Well GW45 is $6.9 \mu\text{g/L}$, which is less than expected from this simple calculation. The fact that the observed concentration is somewhat less than could theoretically be generated by the sediments suggested that the sediments are the predominant source of ground water contamination in this area.

As the contaminated ground water migrates downgradient toward Well GW47, the plume will disperse laterally. The plume width is estimated by the following equation (see App. A for derivation):

$$W_r = W_s + 1.03 X$$

where: W_r = the plume width at the receptor location,

W_s = the plume width at the source, and

X = the source-receptor distance along a limiting flow line.

AR304948



Based on an analysis of Fig. 4.2-1, $X = 176$ m. From Sec. 4.2.1, $W_g = 220$ m; therefore, $W_r = 220 + 1.03(176) = 401$ m. Assuming the aquifer thickness is approximately the same at each location, the plume concentrations will have diluted by the ratio of $W_r:W_g$, or 1.8:1. The estimated ground water concentration of 2,4-DNT at GW47 is thus estimated to be $(35/1.8)$ $\mu\text{g/L}$ to $(500/1.8)$ $\mu\text{g/L}$ or 19 to 280 $\mu\text{g/L}$. The observed 2,4-DNT concentration at Well GW47 is 24.6 $\mu\text{g/L}$, i.e., within the estimated range.

These calculations tend to indicate that the conceptual model expressed in the first paragraph of this section explains the predominant contaminant migration pathway, and that the Red Water Reservoirs sediments are the most important source of contamination in this area. The ability to estimate ground water contaminant levels by this simple calculation suggest that it might be useful for order of magnitude extrapolation to downgradient locations. If the same procedure is used to estimate the concentration at the Schwartz well, approximately 750 m downgradient:

$$\begin{aligned} W_r &= 220 + 1.03(750) \\ &= 990 \text{ m} \end{aligned}$$

$$\begin{aligned} \text{Dilution ratio} &= 990:220 \\ &= 4.5:1 \end{aligned}$$

Estimated 2,4-DNT concentration = 5.6 to 110 $\mu\text{g/L}$

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The observed concentration of 2,4-DNT at the Schwartz well is <0.3 µg/L.
The estimated travel time to the Schwartz well is:

$$\begin{aligned}\text{Travel time} &= \frac{\text{distance}}{\text{flow velocity}} \\ &= \frac{750 \text{ m}}{0.5 \text{ m/day}} \\ &= 1,500 \text{ days} \\ &= 4 \text{ years}\end{aligned}$$

where: flow velocity estimated by ESE (1987b).

Since the Red Water Reservoirs received wastewater more than 40 years ago, sufficient time has passed for transport to the Schwartz well. Because concentrations are less than expected, it is concluded that the main portion of the contaminant plume discharges to Mill Creek east of SR 62.

AR304950

4.3 POND 13/WET WELL AREA

4.3.1 Site Characteristics Affecting Fate/Transport

Pond 13 is one in a series of impoundments discharging to Oldtown Creek. The discharge of Pond 13 can be estimated as follows:

A runoff of 12 in/yr, or 25 percent of precipitation, over a drainage basin of 60 acres, including upgradient Ponds 14 and 15, yields a discharge of 19.4 million gallons per year (MG/yr).

The wet wells are closed basins with negligible drainage areas. With a net precipitation of 14 in/yr and the diameter of each basin approximately 75 ft, annual infiltration is approximately 0.08 MG/yr. Monitor well locations at this area are shown on Fig. 4.3-1.

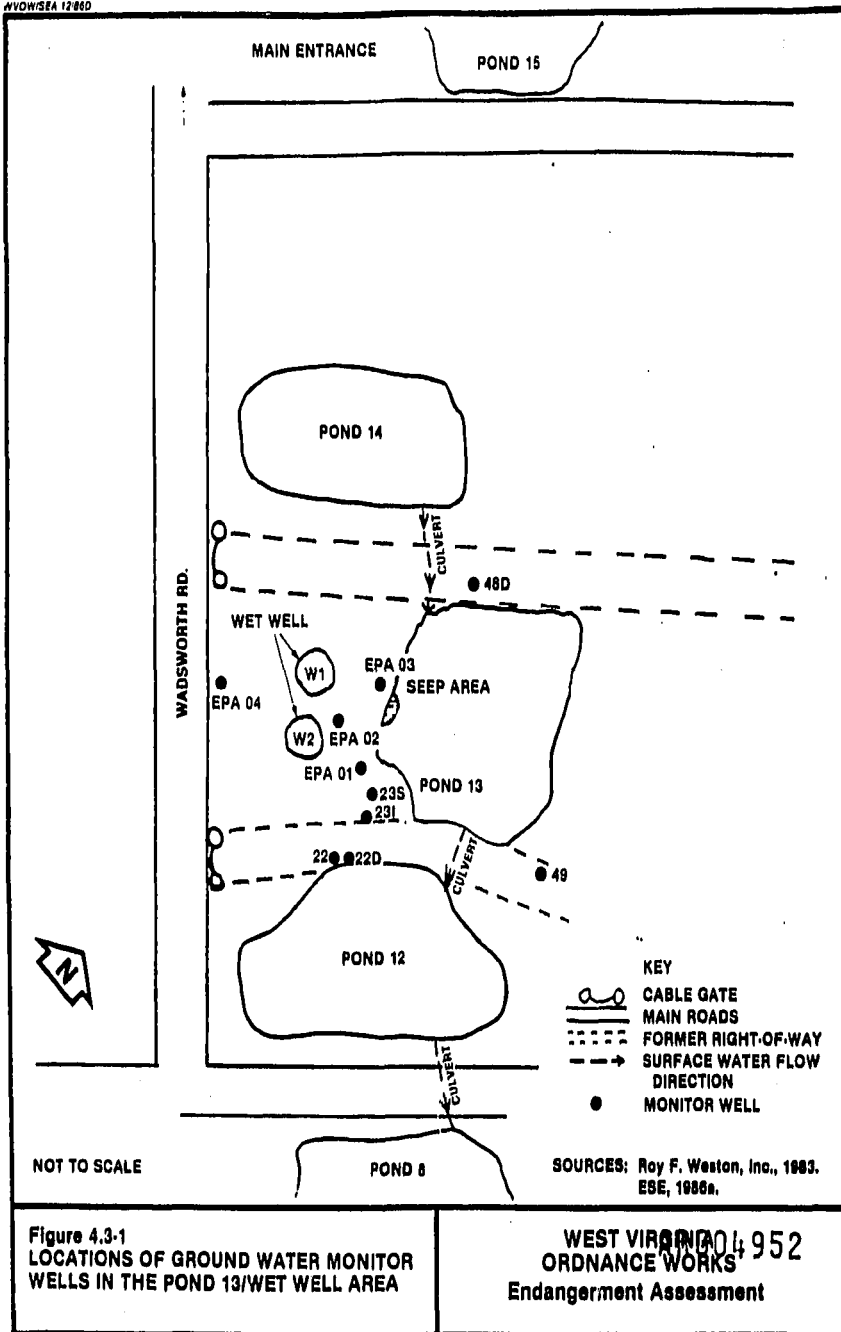
A geologic cross section through the Pond 13/Wet Well Area is shown in Fig. 4.3-2. The lithologic data from monitor wells installed during the field surveys have indicated that two distinct hydrogeologic environments are present in this study area. A shallow sand aquifer of limited areal extent is present; Wells GW23S, GW23I, GW49, and the four EPA wells are completed in this aquifer. The shallow aquifer terminates to the north and to the southeast as evidenced by the substantial clay thicknesses observed at Well GW48D and GW22/GW22D, respectively.

Water levels in the shallow aquifer indicate that little or no contaminant transport occurs in the limited water-bearing unit (Fig. 4.3-3). The shallow aquifer is bounded by vertically and areally extensive clay deposits. The gray clay confining layer is present below the Pond 13 area and is of sufficient thickness to preclude vertical contaminant migration.

4.3.2 Source Area Characterization

The source of contamination in the Pond 13/Wet Well Area is subsurface sediments of the two wet wells, which are associated with a pump station formerly situated in this area.

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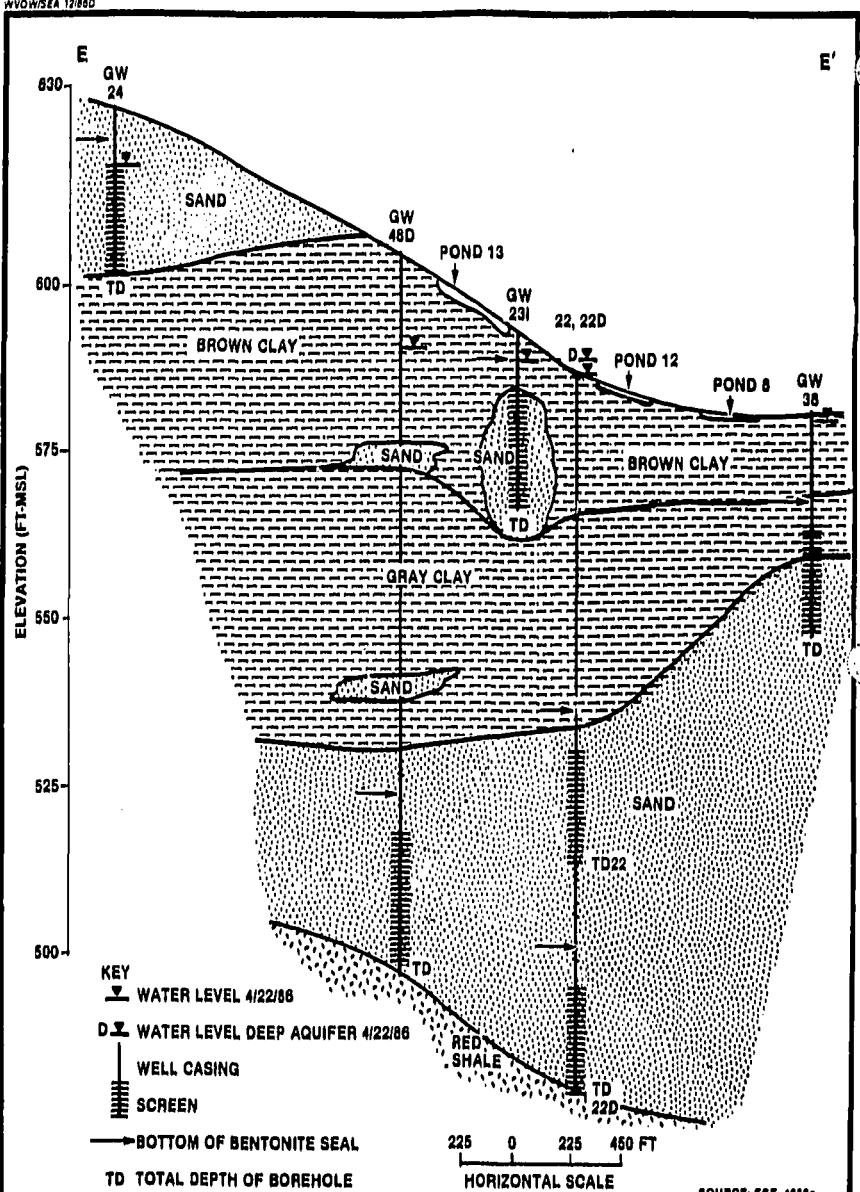
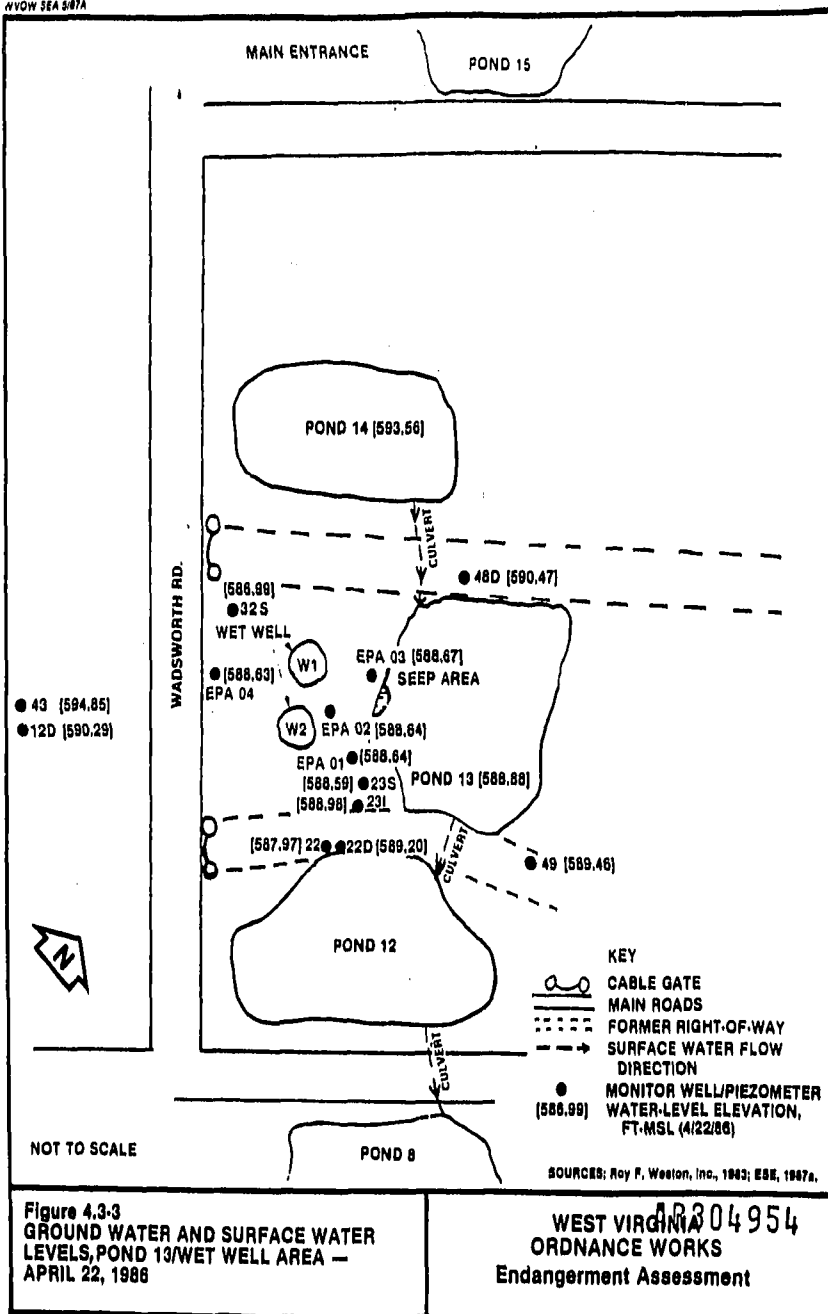


Figure 4.3-2
GEOLOGIC CROSS-SECTION,
POND 13/WET WELL AREA

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Only one of several sediment samples taken from the wet wells was contaminated. This sample was analyzed for 2,4,6-TNT, and the concentration was 4,240 µg/g. Based on ratios of 2,4-DNT to 2,4,6-TNT observed in other subsurface soils and sediments at WVOW, it may be assumed that the level of 2,4-DNT present in that sample was less than 200 µg/g. Surficial sediments in the wet wells are not contaminated so exposure can occur only after leaching and transport via shallow ground water to Pond 13.

4.3.3 Contaminant Migration Pathways

Quantitative analysis of contaminant migration pathways in the Pond 13/Wet Well Area would be highly uncertain for several reasons:

1. Only one sample of wet well sediments was collected from the contaminated zone, making source characterization uncertain.
2. The hydrogeologic system is relatively complex, making quantification of contaminant mass transport via ground water highly uncertain. Nonetheless the ground water flow rate appears to be very slow, and the contaminated shallow aquifer is limited in horizontal extent.
3. Ground water quality in the shallow aquifer varies significantly over short distances, resulting in further uncertainties in quantification of contaminant migration.

From the single sample taken from a contaminated zone in the wet well sediments, the concentration of 2,4,6-TNT in leachate would be estimated to be:

$$C_1 = \frac{4,240 \text{ } \mu\text{g/g}}{0.3 \text{ to } 2.1 \text{ mL/g}}$$

$$= 2,000 \text{ to } 14,000 \text{ } \mu\text{g/mL}$$

$$= 2 \times 10^6 \text{ to } 14 \times 10^6 \text{ } \mu\text{g/L}$$

Note: mL/g = milliliters per gram, and
µg/mL = micrograms per milliliter.

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These estimated values exceed the solubility of 2,4,6-TNT (approximately 1.5×10^5 $\mu\text{g/L}$), indicating that the most contaminated wet well sediments may contain crystalline (precipitated) TNT, rather than adsorbed contaminant.

The observed ground water concentrations [less than 50,000 $\mu\text{g/L}$ total nitroaromatics (ESE, 1986d)] are much less than this level, suggesting that the wet well sediments alone can account for the observed levels in the shallow aquifer and Pond 13. It is possible, due to the slow rate of ground water movement, that steady-state concentrations have not been achieved and that concentrations could increase in the future if no action is taken.

AR304956

5.0 EXPOSURE ASSESSMENT

5.1 ACIDS AREA/YELLOW WATER RESERVOIR AREA

5.1.1 Pathways

Access to contaminated surficial soils in the Acids Area/Yellow Water Reservoir area is unrestricted. The area is not part of McClintic Wildlife Station. The only significant existing exposure pathway is direct contact with these surficial soils. At the hot spot near the neutralization chamber, total nitroaromatic contaminants were observed at 12,200 µg/g (by the field colorimetric method), 2,4,6-TNT at 10,000 µg/g [by laboratory gas chromatography-electron capture (GC-EC) analysis] and 2,4-DNT at <90 µg/g. By analogy to other surficial soil samples collected at WVOW, it can be assumed that the 2,6-DNT concentration is <0.3 percent of total nitroaromatics by the field colorimetric method (i.e., <37 µg/g) (see ESE, 1986b, Sec. 6.6).

5.1.2 Population at Risk

The Acids Area/Yellow Water Reservoir is not an attractive play area for children, and there are no residences within 0.5 mi of the surficial contaminant hot spot. Unlike the McClintic Wildlife Station, recreational use of the area is not encouraged. The land is not used for any purpose at this time. Therefore, it is unlikely that an individual would spend a significant amount of time in this area. The size of the population at risk is unknown.

5.1.3 Exposure Levels

In heavily frequented areas, direct contact has been assumed to result in exposure to the contaminant mass associated with 1.4×10^{-4} kilogram (kg) of soil per day (see ESE, 1986b, Sec. 6.5). Versar (1986) recommends use of a higher exposure factor per event. Using the Versar approach for the same exposure scenario would result in an exposure estimate of 2.3×10^{-3} kg soil/day. The ESE (1986b) approach, based on the work of Hawley (1985), is recommended for this assessment because of absorption efficiency and the fact that effective contact with the skin

surface is not achieved under exposure to soil-absorbed contaminants. This latter effect is referred to as the matrix effect. The Agency for Toxic Substances and Disease Registry (ATSDR) (1987) recommends use of a higher absorption rate than the rate used by Hawley (and thus in this assessment). ATSDR (1987) also concludes there are insufficient data available to define the value of the matrix effect for nitroaromatics. It is recognized that substantial uncertainties exist in any estimate of dermal absorption of soil-absorbed contaminants. Hawley's (1985) recommended absorption rate is higher than reported values for most compounds tested, and a factor of 20 higher than the absorption rate observed for nitrobenzene, the only nitroaromatic tested. Thus it is concluded that the Hawley (1985) analysis is sufficiently conservative. On the other hand, the lack of consideration of these documented factors by Versar (1986) makes its approach excessively and unrealistically conservative. Because of the low utilization of this area, a factor of 100 adjustment is reasonable; therefore, an individual would be exposed to 1.4×10^{-6} kilogram per day (kg/day). Exposure levels are estimated by:

$$\text{dose (mg/kg/day)} = \frac{\text{maximum concentration (mg/kg)} \times 1.4 \times 10^{-6} \text{ kg soil/day}}{70 \text{ kg}}$$

For example:

$$\begin{aligned} 2,6\text{-DNT dose} &= \frac{37 \text{ mg/kg} \times 1.4 \times 10^{-6} \text{ kg soil/day}}{70 \text{ kg}} \\ &= 7.4 \times 10^{-7} \text{ mg/kg/day} \end{aligned}$$

Exposure to 2,4,6-TNT, 2,4-DNT, and 2,6-DNT would be estimated to be 2×10^{-4} mg/kg/day, $<1.8 \times 10^{-6}$ mg/kg/day, and $<7.4 \times 10^{-7}$ mg/kg/day, respectively. These values are conservative estimates of existing levels of exposure.

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In the future, the Acids Area/Yellow Water Reservoir area may be developed for industrial use. This is the most probable future land use (other than its current undeveloped condition). Adjacent land is developed for industrial, commercial, and institutional (fairgrounds) uses or is undeveloped. The parcel is zoned for industrial use and is owned by the Mason County Development Authority, whose traditional objective has been to attract investment for industrial development. Residential development is unlikely. If the Acids Area/Yellow Water Reservoir is developed for industrial use, it can be assumed that workers could be exposed to contaminated soils/sediments. The depth of contamination in the soil profile is not a factor affecting or limiting exposure because onsite activities (construction or landscaping) could expose contaminated subsoils. Exposure levels for workers are developed and documented in Sec. 6.6.

AR304959

5.2 RED WATER RESERVOIRS AREA

5.2.1 Pathways

The Red Water Reservoirs area is within the boundary of McClintic Wildlife Station. Leachate generated by the source area mixes with ground water in the shallow sand aquifer system and migrates beyond the McClintic Wildlife Station boundary. Migration results in potential exposure to existing and future ground water users. The RI data obtained by ESE (1986d, 1987b) do not indicate that any exposure pathways are completed at this time. The Schwartz well, which exhibits very low levels of contamination, is not used currently for potable supply because the Schwartz residence is supplied by the municipal water system. In the future, domestic supply wells could be installed in contaminated portions of the aquifer near the McClintic Wildlife Station boundary.

The ground water/drinking water pathway is the only potential exposure pathway. The contaminated sediments at the source area are not exposed, and surface water is not contaminated, nor is it expected to become contaminated in the future.

5.2.2 Population at Risk

The population at risk is very small--less than five residences are supplied by domestic wells in the potentially affected area at this time. A low rate of residential property development has occurred in the area during the last 10 years, and most, if not all, new residences are served by the municipal water system.

5.2.3 Exposure Levels

No evidence was developed during the RI to suggest that exposure to contamination from the Red Water Reservoirs occurs at this time. Potential future, worst-case exposure levels for hypothetical new wells installed near the McClintic Wildlife Station boundary may be indicated by the observed concentrations in Well GW47, where 2,4-DNT was observed at 24.6 µg/L. Assuming consumption of 2 liters per day (L/day) of tap water and an adult body weight of 70 kg, as appropriate for a human exposure assessment, exposure to 2,4-DNT could, hypothetically, be as high as 7×10^{-4} mg/kg/day.

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5.3 POND 13/WET WELL AREA

5.3.1 Pathways

The exposure pathway of concern in the Pond 13/Wet Well Area results from leachate generation by contaminated sediment in the Wet Wells, transport by ground water in the shallow sand aquifer system, and discharge to Pond 13, where aquatic biota are exposed. Fishermen may be exposed after bioaccumulation of contaminants from water or sediments and by direct contact with contaminated water. Currently, Pond 13 is closed to fishing; therefore, aquatic biota are the only population exposed.

5.3.2 Population at Risk

Although fishing is prohibited at this time, access to Pond 13 is unrestricted. The objective of remedial action in this area would be to reopen Pond 13 to fishermen. It is assumed, therefore, that the human population at risk is composed of fishermen and their families or friends with whom the catch is shared. Records are not kept on fishermen use and harvest at McClintic Wildlife Station (McDonald, 1984). A variety of freshwater biota indigenous to Pond 13 and the stocked fish may also be considered to be at risk.

5.3.3 Exposure Levels

It is assumed that a fisherman harvests 10 days per year from Pond 13 and catches 3 pounds (1b) of edible flesh each day. This 30-lb annual harvest is shared with four people resulting in a daily average fish consumption of 0.01 kg/day. These assumptions are conservative in the context of assumptions adopted by EPA in establishment of surface water quality criteria, wherein fish and seafood consumption combined was assumed to equal 0.0065 kg/day.

Observed concentrations of nitroaromatics in the Pond 13/Wet Well Area range from 0.4 to 2.0 µg/L for 2,4-DNT. The critical contaminant in the Pond 13/Wet Well Area is 2,4-DNT. Bass is a popular prey of sport fishermen in the Pond 13/Wet Well Area. Prior to discovery, AR 304961

contamination, the pond was stocked with bass. The bioconcentration factor for 2,4-DNT in bass was estimated to be 18.8 liters per kilogram (L/kg) (see ESE, 1986b, Sec. 6.3). ATSDR (1987) recommended that this assessment evaluate the potential for bioaccumulation through the food chain of the bass in addition to direct bioconcentration via exposure to contaminated water. A review of the available literature on aquatic bioaccumulation of nitroaromatic contaminants revealed no direct measurements of bioaccumulation or biomagnification. The cumulative effect of contaminated food sources to successive trophic levels can be calculated by equations derived by Thomann (1981). Application of this model, however, requires knowledge of the assimilation efficiency of the contaminant from food, the weight-specific ration for feeding rate, and the elimination rates of the contaminants at each trophic level in addition to the uptake rates. Since many of these data were not available, the model could not be utilized to derive an estimate of the biomagnification factor. The low bioconcentration factors and high elimination rates observed (Based on limited data) for 2,4-DNT suggest, however, that the potential for biomagnification is very low (Liu et al., 1983). In addition, Bruggeman et al. (1981) evaluated the relative bioaccumulation from food and water and concluded that food chain biomagnification is likely to occur for organics with $\log K_{ow} > 5$, i.e., only for contaminants that are much more lipophilic than the nitroaromatic compounds characteristic of WVOW. Thus, the bioconcentration factors used by ESE (1986b) are believed to provide a valid basis for estimating fish tissue concentrations for the EA.

At the observed 2,4-DNT concentration range, the bass tissue concentration is estimated to be:

$$C_f (\mu\text{g/kg}) = 0.4 \text{ to } 2.0 \mu\text{g/L (18.8 L/kg)}$$

$$= 7.5 \text{ to } 38 \mu\text{g/kg}$$

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Note: $\mu\text{g/kg}$ = micrograms per kilogram.

At a fish consumption rate of 0.01 kg/day, the exposure level is estimated to be 0.075 to 0.38 microgram per day ($\mu\text{g/day}$). Assuming an adult body weight of 70 kg, appropriate for assessment of exposure to carcinogens, the estimated daily intake of 2,4-DNT is 1.1×10^{-6} to 5.4×10^{-6} mg/kg/day).

According to Mr. Tom Dotson, wildlife manager at McClintic Wildlife Station, dermal exposure to contaminated water in the Pond 13/Wet Well Area occurs as fishermen land fish or rinse their hands after handling fish or bait. Fishermen do not wade in the pond to fish, discouraged by the muck bottom, unless equipped with hip boots. It is estimated that the hands are actually wet for 0.5 hour per fishing trip to the Pond 13/Wet Well Area, which were previously estimated (ESE, 1986b) to occur once per month. Thus the average exposure period is 0.5 hour per event, and 1 event per 30 days, or 0.017 hr/day. Exposure is estimated by the method presented by Versar (1986):

$$\text{Exposure } (\mu\text{g/day}) = \frac{(D \times A \times C_w \times \text{Flux})}{P_w}$$

where: D = exposure duration (0.017 hr/day),
A = exposed area (840 cm^2 ; Anderson et al., 1984),
 C_w = contaminant concentration (0.4 to $2.0 \mu\text{g/L}$),
Flux = flux rate of water across skin ($0.5 \text{ mg/cm}^2/\text{hr}$; Versar, 1986), and

P_w = density of water (10^6 mg/L).

$$\text{Exposure } (\mu\text{g/day}) = \frac{0.017 \times 0.084 \times (0.4 \text{ to } 2.0) \times 0.5}{10^6}$$

$$= 2.9 \times 10^{-10} \text{ to } 1.4 \times 10^{-9}$$

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Since dermal exposure is so much less than exposure by fish consumption, there is no need to consider it further in the assessment of exposure to surface water contaminants.

AR304964

6.0 EVALUATION OF SAFE LEVEL OF RESIDUAL CONTAMINATION

To support both the EA and the FS, criteria in the form of maximum safe contaminant levels in water, soil, and sediment in each of the areas of concern must be determined. Comparison of observed levels with these criteria will be the substantive basis for recommending the performance of a FS and will provide guidance regarding appropriate action levels for the FS. The rationale for developing these criteria could also support a petition for Alternate Concentration Limits (ACLs) for ground water. The ACLs could become site-specific ground water standards under the provisions of the Resource Conservation and Recovery Act (RCRA).

Criteria have been developed in consideration of all realistic exposure pathways by which people or wildlife may be exposed to the contaminants, including direct exposure, uptake through the food chain, and consumption of water. Criteria development was modeled on the Preliminary Pollutant Limit Value (PPLV) methodology developed at USAHBRDL. The PPLV methodology represents an approach to criteria development based on site-specific exposure and risk assessment techniques.

6.1 REMEDIAL OBJECTIVES

The analyses presented here are based on the following set of objectives for remediation:

1. Upon completion of remediation, McClintic Wildlife Station will be managed as a hunting and fishing area with free access; recreational users should not be exposed directly or through game and fish to contamination levels which pose a significant risk.
2. Ground water at the McClintic Wildlife Station boundary and offsite should be safe to drink, implying a negligible individual lifetime cancer risk of 10^{-5} or 10^{-6} for any individual who may tap that ground water supply in the future.

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Ground water resources that may have become contaminated are not now used for potable supply. The area is served by a municipal supply, and any new residents can be connected to this supply. However, they are not required to, and the ground water resource is a "usable" aquifer.

Overwhelming institutional constraints militate against the use of ground water on McClintic Wildlife Station as a potable supply. McClintic Wildlife Station is expected to remain a wildlife station in perpetuity. The State of West Virginia has the incentive, intent, and authority (barring national emergency, in which case the Army could take control of the property, presumably for industrial use) to maintain the land in this use.

These factors provide a context in which Objective No. 2 could be applied with some degree of flexibility. Although it would be desirable to achieve drinkable ground water quality at McClintic Wildlife Station boundaries, endangerment by the potential ingestion of usable water supplies not now used, in a locale where an alternative supply is readily available, is unlikely and could be prevented by a variety of means, including:

1. Requiring existing or future residents whose supply is threatened to hook up to the available municipal supply, or
2. Expanding the McClintic Wildlife Station to include some or all of the areas within ground water contaminant plumes.

The safe contaminant levels presented here are based on Objectives 1 and 2, and the assumption that McClintic Wildlife Station boundaries remain as they are today.

AR304966

6.2 SCOPE OF CRITERIA DEVELOPMENT AND METHODS

Criteria based on health risks have been developed for soils/sediments in the Acids Area/Yellow Water Reservoir. Criteria are presented for ground water for areas outside the McClintic Wildlife Station boundary. Independent criteria designed for the protection of both human health and aquatic life have been developed for surface water of the Pond 13/Wet Well Area, which is on the McClintic Wildlife Station. RI data support the hypothesis that contamination associated with the Pond 13/Wet Well Area does not migrate off McClintic Wildlife Station in hazardous concentrations. Exposure scenarios appropriate for the mandated use of McClintic Wildlife Station property as a wildlife management area with unrestricted access are applied to derive criteria for the Pond 13/Wet Well Area.

Contamination in these three areas can affect several distinct populations at risk. The reasonable assumption that these populations are independent is adopted. The populations potentially at risk are:

1. Potential future workers at a hypothetical industrial facility that may be constructed at the Acids Area/Yellow Water Reservoir,
2. Existing and/or potential future residents downgradient of contaminant plume in ground water associated with the Red Water Reservoirs,
3. Recreational fishermen or other visitors to the Pond 13/Wet Well Area, and
4. Freshwater aquatic biota (Pond 13/Wet Well Area).

The Acids Area/Yellow Water Reservoir is on a parcel of land contiguous with McClintic Wildlife Station and owned by the Mason County Development Authority. The land currently is not used industrially or for any other purpose. Its location, developed infrastructure, and zoning suggest that future industrial use is a reasonable assumption. Adjacent commercial, institutional, and industrial development make the land unattractive for residential development. It is assumed that industrial use could occur in the future.

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Pertinent to the Red Water Reservoirs, criteria for ground water used as a drinking water supply are presented. These criteria should be met off McClintic Wildlife Station property. The source of ground water contamination in this area is attributed primarily to sediments of the Red Water Reservoirs. These sediments are covered by clean, recent sediments; therefore, there is no potential for direct exposure or exposure through the aquatic food chain. These sediments can contribute to endangerment via only the ground water pathway. As guidance to the FS, criteria have been developed for these sediments such that achieving these criteria would ultimately result in meeting the ground water criteria at the boundary. The only reason for evaluating remedial options addressing the sediments would be to limit the duration of an action involving ground water (i.e., cost-effectiveness). It is not necessary to achieve the sediment criteria to eliminate the endangerment.

In the Pond 13/Wet Well Area, surface water is the environmental medium resulting in exposure to either fishermen or aquatic life. The observed surface water contamination results from ground water discharge, and the ground water contamination in turn appears to be associated with the abandoned Wet Well sediments. Thus, complementary and self-consistent criteria have been developed for the Wet Well sediments, ground water, and surface waters with the objective of effecting acceptable surface water concentrations. Exposure pathways are completed only via the surface water medium; therefore, criteria for Wet Well sediments and ground water are provided as guidance to the FS to support evaluation of cost-effective options. Contamination of sediments and ground water in the Pond 13/Wet Well Area does not result directly in an endangerment. The contaminated sediments are subsurface, and the ground water is not used.

In all cases, remedial alternatives that reduce exposure cost-effectively and interdict contaminant migration at various points in the pathways should be evaluated. For example, ground water remediation at the

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McClintic boundary downgradient of the Red Water Reservoirs could be evaluated with or without remediation of contaminated sediments in the reservoirs. Cost-effectiveness of an alternative would be the principal basis for choosing between two such alternatives since either would remedy the endangerment.

The method used to determine safe levels is known as the PPLV method and is documented by Small (1984). Each significant source-to-receptor pathway is quantified and the effects combined to ensure that an exposed individual will not receive an unacceptably large dose. Intermediate results of the methods are referred to as single pathway preliminary pollutant limit values (SPPPLVs) and represent residual levels of contamination that would be safe if only that single pathway were operating. Several pathways are combined by the following equation:

$$PPLV = \frac{1}{\frac{1}{SPPPLV,1} + \frac{1}{SPPPLV,2} + \dots}$$

By definition, a PPLV is a site-specific criterion indicative of a safe level of residual contamination. An SPPPLV is an intermediate result in the calculation process; it can be used as a criterion if only one exposure pathway is significant. The terms PPLV and criterion can be used interchangeably; the term PPLV is used to emphasize the methodology used to calculate a PPLV criterion.

The organization of the remainder of this section is as follows. Sec. 6.3 presents the SPPPLV for ground water used as a drinking water supply. For populations living downgradient of contaminated areas, this is the only significant exposure pathway and, therefore, is the appropriate criterion for offpost ground water.

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Potential future workers in the Acids Area/Yellow Water Reservoir may be exposed to soil or sediments by direct contact or inhalation of dust resuspended during construction or other industrial activities. Because future industrial activities onsite could involve construction or other earth-moving activities, subsurface contamination in the Acids Area/Yellow Water Reservoir could be exposed; therefore, soils/sediment criteria for this area apply to all soils/sediments regardless of depth (see Sec. 6.4).

Criteria for sediments in the Red Water Reservoirs are designed to meet the ground water criterion offpost and development of that criterion is presented in Sec. 6.5.

In the Pond 13/Wet Well Area, the exposure pathway is completed in surface water. Surface water criteria were developed in the EA for source areas (ESE, 1986b--Table 6.3.1). Appropriate ground water and Wet Well sediment criteria for the Pond 13 Area designed to achieve these surface water criteria are developed and presented in Sec. 6.6.1 and Sec. 6.6.2, respectively, of this report.



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6.3 SPPPLV FOR GROUND WATER AS A DRINKING WATER SUPPLY

The SPPPLV is calculated by:

$$PPLV(DW) = SPPPLV(DW) = \frac{B_w}{W_w} \times D_c$$

where: SPPPLV(DW) = drinking water criterion (mg/L),

B_w = human body weight (70 kg),

W_w = amount of water ingested daily [2 liters (L)],

and

D_c = acceptable daily intake (mg/kg/day).

For carcinogens, D_c is based on the acceptable risk factor. If a 10^{-6} individual lifetime cancer risk is considered acceptable, the acceptable daily intake is calculated:

$$D_c = \frac{10^{-6}}{q_1^*}$$

where: q_1^* = unit cancer risk.

If a 10^{-5} individual lifetime cancer risk is considered acceptable, the acceptable daily intake is calculated:

$$D_c = \frac{10^{-5}}{q_1^*}$$

Results for drinking water are presented in Table 6.3-1.

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Table 6.3-1. Criteria for Drinking Water

Compound	D_t (mg/kg/day)	SPPLV(DW) (mg/L)	Detection Limit (mg/L)
TNT	1.4×10^{-3}	0.05	8×10^{-5}
DNB	4×10^{-4}	0.014	2×10^{-4}
TNB	5.8×10^{-3}	0.20	2×10^{-3}
2,4-DNT			
10^{-6} Risk	3.2×10^{-6}	1.1×10^{-4}	3×10^{-4}
10^{-5} Risk	3.2×10^{-5}	1.1×10^{-3}	
2,6-DNT			
10^{-6} Risk	6.4×10^{-7}	2.2×10^{-5}	8×10^{-4}
10^{-5} Risk	6.4×10^{-6}	2.2×10^{-4}	

Note that the 10^{-6} risk level is below achievable detection limits for the DNTs.

Source: ESE, 1987a.

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6.4 ACIDS AREA/YELLOW WATER RESERVOIR

It is assumed that the contaminated area will be developed for industrial purposes, consistent with adjacent land use, infrastructure, zoning, intent of the Mason County Development Authority, and acknowledgement of contaminated status in the deed for the parcel. In an industrial land use, human exposure to contaminated soils can occur as a result of direct contact or via inhalation of dust raised by heavy equipment, truck traffic, or earth-moving. Because of the potential for disturbance during construction or landscaping, contaminated subsurface sediments may be exposed; therefore, the criteria presented apply to soils/sediments regardless of depth. Exposure to ground water is disregarded because an industrial water supply would tap the uncontaminated deep sand and gravel aquifer. The contamination affects only the shallow sand aquifer which is not used for water supply at any location in the vicinity of WVOW. Although of sufficient yield to supply a domestic well, the shallow aquifer would not provide adequate supply for industrial use, the anticipated future land use in this area. Because the uncontaminated deep sand and gravel aquifer is the source for all potable supplies (municipal and domestic) in the vicinity, it is assumed that this aquifer would also be used by any future water users. The deep sand and gravel aquifer is well protected from surface contamination by the dense gray clay confining layer, which is more than 20 ft thick at the Acids Area/Yellow Water Reservoir. Furthermore, municipal water supply is available to, and used in, the area. Thus, contamination of the shallow sand aquifer does not pose any endangerment, and health risk-based criteria are not developed for soils/sediments in the Acids Area/Yellow Water Reservoir based on their potential to contaminate the aquifer.

The PPLV criterion is calculated by the following equation:

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$$\text{PPLV (industrial, mg/kg)} = D_c \text{ (mg/kg/day)} \left(\frac{B_w}{W_{s,d} f_d f_e} \right)$$

where: $W_{s,d}$ = ingestion and inhalation of dust during a dusty workday (kg/day),

f_d = dusty workdays as a fraction of the year,

f_e = period of employment as a fraction of 70-year lifetime ($f_e = 1$ for noncarcinogens),

$W_{s,d}$ = incidental ingestion of dust from soiled hands plus the product of total suspended particulates (TSP) x inhalation rate (m^3/day), and

TSP = TSP concentration in the work area [kilograms per cubic meter (kg/m^3)].

Soil can be ingested inadvertently during smoking or eating if hands are dirty. Hawley (1985) reviews and evaluates the limited data available to support quantification of this exposure pathway. Based on review of these data, in particular the most detailed study in which exposure by pesticide applicator personnel was monitored, the incidental ingestion rate was estimated to be $1.1 \times 10^{-4} \text{ kg/day}$, and is based on an assumption that seven cigarettes are smoked during the workday.

The average TSP concentration in the work area is estimated to be $10^{-5} \text{ kg}/\text{m}^3$. This estimate is judgmental, with higher levels (4×10^{-5} to $2 \times 10^{-4} \text{ kg}/\text{m}^3$) observed in association with heavy vehicular traffic. The high reported values must not represent long-term average conditions because vision would be obscured at such high levels. OSHA regulations require the use of respirators at dust levels exceeding $1.5 \times 10^{-5} \text{ kg}/\text{m}^3$. Thus, $1.5 \times 10^{-5} \text{ kg}/\text{m}^3$ is an upper limit for exposure, and a reasonably conservative average exposure level is estimated to be $10^{-5} \text{ kg}/\text{m}^3$.

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Inhalation rate data have been reviewed by Small (1984). Light activity (e.g., office work, laboratory work, hospital work, or housework) ranges up to 14.9 cubic meters (m^3) per 8-hour workday. Heavy activity (commercial fishing, foundry work, face work in mines, or postal delivery on foot) results in an inhalation rate of 20.6 m^3 or more per workday. An intermediate value of 18 m^3 /day is assumed for a variety of industrial activities.

The aerosol retention efficiency of 75 percent is suggested by Hawley (1985).

Applying these values,

$$\begin{aligned}W_{a,d} &= 1.10 \times 10^{-4} \text{ kg/day} + 10^{-5} \text{ kg/m}^3 (18 \text{ m}^3/\text{day}) 0.75 \\&= 2.45 \times 10^{-4} \text{ kg/day}\end{aligned}$$

Dusty workdays are estimated as on the job days with snow cover less than 1 inch and precipitation less than 0.1 inch. On-the-job days are defined as 365 days minus 104 weekend days, 7 holidays, 10 vacation days, and 10 sick days, resulting in 234 on-the-job days. Five years (1981 through 1985) of monthly Local Climatological Data summary sheets were obtained for the National Weather Service station in Huntington, WV. The average number of dusty days (snow cover less than 1 inch and precipitation less than 0.1 inch) was 259 per year. Because dusty days are as likely to occur on workdays as on non-workdays, the dusty workdays are estimated as $234(259/365) = 166$. Thus, $f_d = 166/365 = 0.45$.

Finally, it is a reasonably conservative assumption that the period of employment at the hypothetical Acids Area/Yellow Water Reservoir industrial site is 15 years, resulting in $f_e = 15/70 = 0.21$ for carcinogens and $f_e = 1$ for noncarcinogens. Combining terms, the PPLVs are determined as follows:

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For carcinogens:

$$PPLV \text{ (mg/kg)} = D_c \text{ (mg/kg/day)} 3.02 \times 10^6$$

For noncarcinogens:

$$PPLV \text{ (mg/kg)} = D_c \text{ (mg/kg/day)} 6.35 \times 10^5$$

Criteria for the Acids Area/Yellow Water Reservoir are presented in Table 6.4-1. The criteria for total nitroaromatics in soil by the field colorimetric method were derived by procedures originally presented by ESE (1986b; see Sec. 6.6). They are based on the assumption that the ratio of individual nitroaromatic contaminants as determined by the laboratory-based GC-EC procedure to total nitroaromatics by the field colorimetric analytical technique is relatively constant in soils throughout the site. In subsurface soil and sewerline samples collected and analyzed by both methods (ESE, 1986d), 2,4,6-TNT by the GC-EC method ranged from 0.27 to 1.38 of total nitroaromatics as determined by the field colorimetric method; 2,4-DNT ranged from 0.0006 to 0.05 of total nitroaromatics; 2,6-DNT was 0.00008 of total nitroaromatics in the only sample in which both 2,6-DNT and total nitroaromatics were detectable; whereas 1,3,5-TNB and 1,3-DNB were 1.25 and 0.03 times total nitroaromatics in the single sample in which both were detectable. Since there are relatively few data available to determine the characteristic ratio of individual compounds to total nitroaromatics, the conservative assumption that any soil sample could contain any compound at the maximum ratio observed is adopted. Thus, it is assumed that 2,4,6-TNT is 1.38 times total nitroaromatics; 2,4-DNT is 0.05 of total nitroaromatics; 2,6-DNT is 0.00008 of total nitroaromatics; 1,3,5-TNB is 1.25 times total nitroaromatics; and 1,3-DNB is 0.03 of the total nitroaromatics. A soil containing 200 mg/kg of total nitroaromatics is acceptable at the 10^{-6} risk level since it is inferred that its 2,4,6-TNT concentration is 276 (less than 890); 1,3-DNB is 6 (less than 250); 1,3,5-TNB is 250 (less than 3,700); 2,4-DNT is 10 (at the 10^{-6} criterion); and 2,6-DNT would be 0.016 mg/kg (significantly less than the 10^{-6} risk level of 2 mg/kg).

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Table 6.4-1. PPLV Criteria for Soils/Sediments in the Acids Area/Yellow Water Reservoir

Compound	D_t	PPLV (mg/kg)
2,4,6-TNT	1.4×10^{-3}	890
1,3-DNB	4×10^{-4}	250
1,3,5-TNB	5.8×10^{-3}	3,700
2,4-DNT		
10^{-6} Risk	3.2×10^{-6}	10
10^{-5} Risk	3.2×10^{-5}	100
2,6-DNT		
10^{-6} Risk	6.4×10^{-7}	2
10^{-5} Risk	6.4×10^{-6}	20
Total Nitroaromatics		
10^{-6} Risk	NA	200
10^{-5} Risk	NA	650

NA = Not applicable.

Source: ESE, 1987b.

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6.5 RED WATER RESERVOIRS

Sediments of the Red Water Reservoirs on McClintic Wildlife Station are contaminated. An acceptable sediment concentration is derived from the following equation:

$$PPLV(\text{sediments}) = SPPLV(\text{sediments to drinking water})$$

$$= \frac{B_w}{W_w} \times D_t \times (K_{sw} \times \text{dilution})^{-1}$$

where: B_w = human body weight (70 kg),

W_w = amount of water consumed daily (2 L/day),

K_{sw} = partition coefficient between sediment and water
[kilogram per liter (kg/L)], and

dilution = dilution factor between sediment interstitial water
(leachate) and ground water at the nearest point of
potential exposure (dimensionless).

In the equation, the term $(K_{sw} \times \text{dilution})^{-1}$ is the ratio between the concentration in ground water at the point of withdrawal and the concentration in sediments of the source area.

K_{sw} as used in this equation is given by:

$$K_{sw} = 1/K_d$$

The K_d values for nitroaromatic compounds in subsoils are given in Table 2.1-3.

The dilution factor is the ratio of contaminated leachate flow through the contaminated soils to the volume of ground water flow at the point of exposure. This factor accounts for the fact that the water withdrawn from water supply wells is not pure leachate, but a mixture of

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leachate with uncontaminated ground water. This ground water underflows the contaminated area or enters the aquifer downgradient of the contaminated area before the point of withdrawal. The dilution factor has been calculated based on Darcy's law, the observed permeability of soils and aquifer material, aquifer thickness, and the observed hydraulic gradients.

At the Red Water Reservoirs, the point of maximum potential exposure by the ground water pathway is downgradient approximately 700 ft at the northwest corner of the McClintic Wildlife Station. Along this pathway, any leachate from the reservoir would mix with uncontaminated ground water prior to withdrawal. For the Red Water Reservoirs, the dilution factor is given by:

$$\text{dilution} = \frac{(P - E) \times A}{K_h \times A_c \times i}$$

where: $(P - E)$ = net precipitation = precipitation minus pond evaporation [14 in/yr = 1.13×10^{-6} centimeters per second (cm/sec)];

A = the surface area of the reservoirs [3.5×10^8 square centimeters (cm²)];

K_h = the horizontal permeability in the aquifer (7.3×10^{-3} cm/sec, ESE, 1987b);

A_c = the cross-sectional area of the potentially contaminated portion of the aquifer at the nearest point of potential exposure; and

i = the hydraulic gradient in the aquifer (0.02, ESE, 1987b).

The cross-sectional area is estimated by considering dispersion along the transport pathway based on procedures presented in App. A. Leachate can enter the aquifer across a width (transverse to limiting flow) of 100 ft.

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of 220 m. Applying relationships presented in App. A, the width of the plume at the McClintic Wildlife Station boundary is estimated to be 440 m. The plume is estimated to have dispersed uniformly through the full aquifer thickness of 5 m. Thus $A_c = 2.2 \times 10^7 \text{ cm}^2$ at the boundary, and the dilution factor is:

$$\text{dilution} = \frac{1.13 \times 10^{-6} \times 3.5 \times 10^8}{7.3 \times 10^{-3} \times 2.2 \times 10^7 \times 0.02} = 0.12$$

This dilution factor is corroborated by data collected in RI (refer to Sec. 4.2).

The dilution factor indicates that each gallon of water that passes through contaminated sediments in the Red Water Reservoirs is mixed with approximately 8 gallons (gal) of uncontaminated ground water prior to reaching the nearest point of potential exposure. Results of the sediment criterion calculation are presented in Table 6.5-1. The derivation of a total nitroaromatic criterion is presented in ESE (1986b) and discussed also in Sec. 6.4 of this report.

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Table 6.5-1. Criteria for Sediments of the Red Water Reservoirs

Compound	D_t (mg/kg/day)	K_{sw}	PPLV (mg/kg)
TNT	1.4×10^{-3}	1	0.4
DNB	4×10^{-4}	5	0.02
TNB	5.8×10^{-3}	1	1.7
2,4-DNT			
10^{-6} Risk	3.2×10^{-6}	3	0.0003
10^{-5} Risk	3.2×10^{-5}		0.003
2,6-DNT			
10^{-6} Risk	6.4×10^{-7}	3	0.00006
10^{-5} Risk	6.4×10^{-6}		0.0006
Total Nitroaromatics			
10^{-6} Risk	NA	NA	0.006
10^{-5} Risk	NA	NA	0.06

NA = not applicable.

Source: ESE, 1987a.

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6.6 POND 13/WET WELL AREA

In the Pond 13/Wet Well Area, the only significant exposure pathways are completed upon discharge of contaminated ground water to Pond 13. Aquatic biota and recreational fishermen are the populations at risk. Criteria to protect these populations were presented by ESE (1986b) and are repeated in Table 6.6-1.

6.6.1 Criteria for Ground Water in the Pond 13/Wet Well Area

In the Pond 13 Area, contaminated ground water discharges to Pond 13 and results in exceedances of the surface water criteria presented in Table 6.6-1. No exposure to contaminated ground water is possible prior to this discharge, so the surface water criteria are the basis for ground water criteria in this area, accounting for the dilution that occurs upon discharge. Thus, the PPLV for ground water is determined by the following equation:

$$\text{PPLV(GW)} = \frac{\text{PPLV(SW)}}{\text{dilution}}$$

$$\text{where: dilution} = \frac{\text{contaminated ground water discharge to Pond 13}}{\text{Pond 13 discharge}}$$

$$\text{Pond 13 discharge} = 19.4 \text{ MG/yr (refer to Sec. 4.3.1)}$$

ATSDR (1987) recommended that the PPLV for surface waters at WVOW be reevaluated to take into account the potential for bioaccumulation of nitroaromatics through the aquatic food chain and the potential for direct dermal exposure to contaminated water. As presented in Section 5.3.3, bioaccumulation of nitroaromatic contamination through the aquatic food chain cannot be estimated from available data, but is not expected to be significant for nitroaromatic compounds. Further, it was also shown in Section 5.3.3 that exposure via dermal absorption is negligible in comparison with consumption of fish. Thus, AR304982 water PPLVs derived by ESE (1986b) are sufficiently conservative remedial objectives for the Pond 13/Wet Well Area.

Table 6.6-1. Criteria for Pond Waters

Compound	PPLV ($\mu\text{g/L}$)
TNT	60
DNB	160
TNB	80
2,4-DNT	
10 ⁻⁶ Risk	3.4
10 ⁻⁵ Risk	34
2,6-DNT	
10 ⁻⁶ Risk	0.67
10 ⁻⁵ Risk	6.7

Source: ESE, 1986a.

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As discussed in Sec. 4.3.1, the geohydrology of the contaminated shallow aquifer system in the Pond 13/Wet Well Area is relatively complex. The aquifer is limited in horizontal extent, and the ground water flow pattern is indistinct and apparently quite sluggish. The only basis for estimating the discharge is from a water-balance perspective, i.e.,

$$\text{ground water discharge} = \text{infiltration} \times \text{recharge area}$$

From geologic and topographic data, the recharge area contributing to ground water discharge to Pond 13 is estimated to be 150,000 ft² (3.4 acres). Infiltration through the Wet Wells, which are closed basins (8,800 ft² in area), is estimated to equal net precipitation (14 in/yr., see Sec. 6.6.2), whereas infiltration through the remainder of the recharge area is estimated to be 2 in/yr (see ESE, 1986b) over an area of 140,000 ft².

Thus:

$$\begin{aligned}\text{ground water discharge ft}^3/\text{yr} &= (1.2 \times 8,800) + (0.17 \times 140,000) \\ &= 34,000 \text{ ft}^3/\text{yr} \\ &= 0.25 \text{ MG/yr}\end{aligned}$$

Note: ft³/yr = cubic feet per year.

Therefore:

$$\begin{aligned}\text{dilution} &= \frac{0.25 \text{ MG/yr}}{19.4 \text{ MG/yr}} \\ &= 0.013.\end{aligned}$$

The PPLV criteria for ground water are presented in Table 6.6-2.

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Table 6.6-2. Ground Water Criteria for Pond 13/Wet Well Area

Compound	PPLV ($\mu\text{g/L}$)
TNT	4,600
DNB	12,000
TNB	6,200
2,4-DNT	
10^{-6} Risk	260
10^{-5} Risk	2,600
2,6-DNT	
10^{-6} Risk	52
10^{-5} Risk	520

Source: ESE, 1986a.

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6.6.2 Wet Well Sediments

Sediments in the two Wet Wells are apparently the predominant source of contaminants to the shallow aquifer in the Pond 13/Wet Well Area and ultimately to Pond 13. The contaminated sediments are isolated from the surface by more recent, uncontaminated deposits, so the only potential exposure pathway is through the surface water discharge. Thus, the criterion for Wet Well sediments is given by:

$$\text{PPLV(Wet Well sediments)} = \frac{\text{PPLV(SW)}}{(K_{aw} \times \text{dilution})}$$

The K_{aw} values are given in Table 6.6-1.

$$\text{dilution} = \frac{(P - E) \times (\text{Wet-Well area})}{\text{Pond 13 discharge}}$$



$$\begin{aligned} P - E &= \text{precipitation minus evaporation} \\ &= 14 \text{ in/yr} \\ &= 8.7 \times 10^{-6} \text{ million gallons per square foot per year} \\ &= (\text{MG/ft}^2/\text{yr}) \text{ (ESE, 1986d)} \end{aligned}$$

$$\begin{aligned} \text{Wet Well area} &= 2 \times 3.14 \times (37.5 \text{ ft})^2 \\ &= 8.8 \times 10^3 \text{ ft}^2 \end{aligned}$$

Therefore:

$$\begin{aligned} \text{dilution} &= \frac{8.7 \times 10^{-6} (8.8 \times 10^3)}{19.4} \\ &= 4.0 \times 10^{-3} \end{aligned}$$

The PPLV calculations for Wet Well sediments are summarized in Table 6.6-3. The procedure for developing the total nitroaromatic criterion is presented in ESE (1986b) and discussed in Sec. 6.4 of this report.

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Table 6.6-3. Criteria for Pond 13/Wet Well Area Sediments

Compound	K _{sw} (kg/L)	PPLV(SW) (µg/L)	PPLV for Wet Well Sediments (mg/kg)
TNT	1	60	15
DNB	5	160	8.0
TNB	1	80	20
2,4-DNT			
10 ⁻⁶ Risk	3	3.4	0.28
10 ⁻⁵ Risk	3	34	2.8
2,6-DNT			
10 ⁻⁶ Risk	3	0.67	0.056
10 ⁻⁵ Risk	3	6.7	0.56
Total Nitroaromatics			
10 ⁻⁶ Risk	NA	NA	5.6
10 ⁻⁵ Risk	NA	NA	11

NA = not applicable.

Source: ESE, 1986a.

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7.0 RISK AND IMPACT EVALUATION

The objective of this risk characterization is to integrate information developed in the toxicity assessment (Sec. 3.0) and the exposure assessment (Sec. 5.0) into a complete evaluation of potential or actual risk. The risk evaluation addresses:

1. Carcinogenic risks,
2. Noncarcinogenic risks
3. Environmental risks, and
4. Risks to public safety and welfare for contaminants and exposure pathways relevant to WVOW.

7.1 ACIDS AREA/YELLOW WATER RESERVOIR

Based on the findings of the RI (ESE, 1986d, 1987b) and previous sections of this report, the no-action alternative is not viable in this area. Surficial soils at a small hot spot of contamination adjacent to the neutralization facility exhibit contamination by 2,4,6-TNT in excess of the criterion developed to protect the health of workers, assuming the land is developed for industrial use in the future. Furthermore, subsurface sediments from the yellow water sewerline and one sample from the Yellow Water Reservoir also exceed these criteria. Exposure to these contaminants under the existing land use is believed to be very low to negligible, so the endangerment to human health relates only to potential future uses. Vegetation stress (bare ground) was also observed at the hot spot.

7.1.1 Human Health

7.1.1.1 Carcinogenic Risks

The compounds 2,4-DNT and 2,6-DNT are the only animal carcinogens of potential concern in the Acids Area/Yellow Water Reservoir.

The carcinogenic risk to potential future workers at a hypothetical industrial facility in this area is estimated with respect to the hot spot of surficial contamination. Subsurface soils/sediments are also

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contaminated at lower levels, and this contamination could be exposed during industrial use. The maximum observed concentrations of 2,4-DNT and 2,6-DNT at the neutralization facility were 0.279 µg/g (0.279 mg/kg) and 0.185 µg/g, respectively. One sample of surficial soils in this area, however, exhibited 12,200 µg/g of total nitroaromatics (principally 2,4,6-TNT), and the high concentration of this contaminant resulted in high detection limits in that sample for the DNTs. Based on observed ratios between the various nitroaromatics in surficial soil samples from WVOW, it is likely that this sample contained DNTs at higher levels than those detected in the sample from the adjacent area. The 2,4-DNT concentration was probably less than 90 mg/kg, and the 2,6-DNT concentration was probably less than 37 mg/kg.

Under the exposure assumptions presented in Sec. 5.1, exposure to individuals under the existing land use is probably less than 1.8×10^{-6} mg/kg/day of 2,4-DNT and 7.4×10^{-7} mg/kg/day of 2,6-DNT. The carcinogenic risk resulting from exposure to 2,4-DNT under the existing land use can be calculated by:

$$\text{Risk} < 10^{-6} \times \frac{\text{estimated exposure}}{\text{exposure resulting in less than } 10^{-6} \text{ risk}}$$

$$< 10^{-6} \times \frac{1.8 \times 10^{-6}}{3.2 \times 10^{-6}}$$

$$< 5.6 \times 10^{-7}$$

The carcinogenic risk associated with exposure to 2,6-DNT under the existing land use is estimated as:

$$\text{Risk} < 10^{-6} \times \frac{7.4 \times 10^{-7}}{6.4 \times 10^{-7}}$$

$$< 1.2 \times 10^{-6}$$

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Summary of Carcinogenic Risks, Existing Land Use--Assuming that carcinogenic risks associated with 2,4-DNT and 2,6-DNT are additive (ICF, Incorporated, 1985), the overall lifetime carcinogenic risk to individuals who are exposed infrequently to surficial soils in the Acids Area/Yellow Water Reservoir is estimated to be less than 1.8×10^{-6} .

Carcinogenic risks for potential future workers at the site under the potential future industrial land use can be estimated by comparison of maximum contaminant levels with the criterion for industrial land use developed in Sec. 6.4. The risk to this hypothetical population is estimated by:

$$\text{Risk} < 10^{-6} \frac{\text{maximum concentration level}}{\text{PPLV criterion at } 10^{-6} \text{ risk}}$$

Risks associated with potential future exposure to 2,4-DNT are, thus:

$$\text{Risk} < 10^{-6} \frac{90 \text{ mg/kg}}{10 \text{ mg/kg}}$$

$$< 9 \times 10^{-6}$$

Risks associated with potential future exposure levels to 2,6-DNT are estimated to be:

$$\text{Risk} < 10^{-6} \frac{37 \text{ mg/kg}}{2 \text{ mg/kg}}$$

$$< 1.9 \times 10^{-5}$$

Summary of Carcinogenic Risks to Potential Future Workers Under an Industrial Land Use--Assuming that the carcinogenic risk associated with exposure to 2,4-DNT and 2,6-DNT are additive (ICF, Incorporated, 1985),

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the overall lifetime exposure risk to potential future workers at a hypothetical industrial facility sited at the Acids Area/Yellow Water Reservoir is less than 2.8×10^{-5} .

7.1.1.2 Noncarcinogenic Risks

Health risks for noncarcinogenic contaminants can be characterized by comparing expected exposure levels to acceptable intake levels [ADI = acceptable intakes, chronic (AIC) = D_t]. The resulting hazard index (HI) provides a relative measure of the spread between acceptable and unacceptable exposure levels. When the exposure level exceeds the ADI ($HI > 1$), the potential exists for adverse health effects for exposed populations. Estimation of exposure levels is performed according to the same methods and assumptions used for carcinogens.

2,4,6-TNT--The maximum observed 2,4,6-TNT concentration in surficial soils at the neutralization facility hot spot is 10,600 mg/kg. The estimated exposure level under the existing land use is 2×10^{-4} mg/kg/day (see Sec. 5.1). Thus, the hazard index is:

$$HI = \frac{\text{estimated exposure level}}{D_t}$$

$$= \frac{2 \times 10^{-4}}{1.4 \times 10^{-3}}$$

$$= 0.14$$

Since the hazard index is much less than 1, no adverse health effects associated with exposure to 2,4,6-TNT are expected under the existing land use.

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In the future, the Acids Area/Yellow Water Reservoir may be developed for industrial use. The hazard index under this land use is estimated by comparing the maximum observed concentration with the PPLV criteria presented in Table 6.4-1. For 2,4,6-TNT, the hazard index for the potential future land use is:

$$HI = \frac{10,600}{4,000}$$

$$= 2.7$$

Since the hazard index exceeds 1, the potential exists for adverse health effects associated with exposure to 2,4,6-TNT under the hypothetical future land use.

1,3-DNB--1,3-DNB was not detected in surficial soils in the Acids Area/Yellow Water Reservoir with a maximum detection limit of 160 mg/kg. Since the PPLV criterion is 1,200 mg/kg (Table 6.4-1), the hazard index is much less than 1, and no adverse health effects are expected under either the existing land use or the potential future land use as a result of exposure to 1,3-DNB.



1,3,5-TNB--1,3,5-TNB was not detected in the Acids Area/Yellow Water Reservoir with a maximum detection limit of 417 µg/g (417 mg/kg). Since the PPLV criterion for this area is 18,000 mg/kg (Table 6.4-1), the hazard index is much less than 1, and no adverse health effects associated with exposure to 1,3,5-TNB are expected under either the existing or potential future land uses.

Lead--As discussed in Section 3.5.4, a quantitative dose-response relationship for lead is not well established. Young children are the population at greatest risk with respect to lead exposure, and neither the current or probable future land use results in significant exposure of children to soil contaminants in this area. The proposed remedial

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maximum contaminant level (RMCL) for lead (EPA, 1985b) implies an allowable exposure by the drinking water pathway of 20 ug/day for young children who are assumed to consume 1 L of water per day. Based on research results by Binder et al. (n.d.), incidental soil ingestion by children is approximately 250 mg/day. At the maximum observed soil concentration of 100 mg/kg, lead exposure would be 25 ug/day, which is approximately equal to the allowable exposure if the area were an active play area. The area is not used as a play area under the existing or probable future land use, so the observed levels of contamination do not pose an unacceptable risk.

Summary of Noncarcinogenic Risks--No adverse health effects are expected associated with noncarcinogenic contaminants in this area under the existing land use. Under potential future industrial land use, workers may be exposed to unacceptable levels of 2,4,6-TNT.

7.1.2 Environmental Impacts

Vegetation stress (bare ground) was observed in the hot spot of surficial soil contamination adjacent to the neutralization chamber. In general, vegetation stress was observed at WVOW when surficial soils approached or exceeded 1,000 µg/g of total nitroaromatics. Utilization of this area by terrestrial biota has not been documented. It is not part of the McClintic Wildlife Station.

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7.2 RED WATER RESERVOIRS AREA

Based on the findings of the RI and preceding sections of the EA, no population is currently exposed to unacceptable exposure levels of contamination associated with the Red Water Reservoirs area. The only potable well contaminated by nitroaromatics (Schwartz well) related to ordnance production is affected by the Red Water Reservoirs. This well is not currently in use, however, because the Schwartz residence is supplied by the municipal water system. Moreover, the contaminant levels observed (1,3-DNB at 0.9 µg/L and 2,4,6-TNT at 0.2 µg/L) are more than an order of magnitude below the criteria defining acceptable levels of contamination in drinking water (see Table 6.3-1); therefore, this water could be used safely. Contaminant travel times to the Schwartz well and other residences in that area indicate that ample time has passed since the Red Water Reservoirs began receiving wastes to have allowed ground water to achieve a steady-state response. Higher concentrations in these wells are not expected to occur. The risks associated with the Red Water Reservoirs area relate only to the potential that, in the future, domestic water supply wells might be installed in areas nearer the McClintic Wildlife Station boundary, where contaminant levels are higher and of unacceptable quality. If this hypothetical domestic well were located outside the McClintic Wildlife Station boundary and immediately downgradient of GW47, it would be expected to have similar quality to that well. GW47 exhibited 24.6 µg/L of 2,4-DNT, 5.4 µg/L of 2,4,6-TNT, and 2.3 µg/L of 1,3-DNB.



7.2.1 Human Health

7.2.1.1 Carcinogenic Risks

The compounds 2,4-DNT and 2,6-DNT are the only animal carcinogens of potential concern in the Red Water Reservoirs area.

The carcinogenic risk to potential future residents using shallow ground water as a source of domestic water supply is estimated by assuming the water has the same quality as Monitor Well GW47 at the McClintic Wildlife

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Station boundary. As a result of dispersion and dilution, actual concentrations probably would be less than this level at potential well locations farther downgradient. GW47 contained 24.6 $\mu\text{g/L}$ of 2,4-DNT and less than 8 $\mu\text{g/L}$ of 2,6-DNT.

The exposure level to 2,4-DNT is given by:

$$\begin{aligned}\text{Exposure} &= \frac{24.6 \mu\text{g/L} \times 2 \text{ L/day}}{70 \text{ kg}} = 0.70 \mu\text{g/kg/day} \\ &= 7.0 \times 10^{-4} \text{ mg/kg/day}\end{aligned}$$

Note: $\mu\text{g/kg/day}$ = microgram per kilogram per day.

The carcinogenic risk associated with potential future use of contaminated ground water is calculated by comparing the estimated exposure level with the dose corresponding to a given level of risk (see Sec. 3.2.4):

$$\text{Risk} < 10^{-6} \times \frac{\text{estimated exposure}}{\text{exposure resulting in less than } 10^{-6} \text{ risk}}$$

$$< 10^{-6} \times \frac{7.0 \times 10^{-4}}{3.2 \times 10^{-6}}$$

$$< 2.2 \times 10^{-4}$$

Because this is the only exposure pathway for the population at risk, the overall risk associated with the potential future consumption of ground water contaminated by 2,4-DNT is less than 2.2×10^{-4} . This risk level is outside the acceptable risk range recommended by EPA; the No-Action Alternative is not viable for the Red Water Reservoirs area.

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2,6-DNT was not detected in ground water at GW47 at a detection limit of 8 µg/L. This detection level is higher than that normally achieved in other samples, resulting from dilution of the sample required to quantify the relatively high level of 2,4-DNT observed in this sample. 2,6-DNT was observed at other locations upgradient of GW47 (e.g., GW45 and GW30); therefore, it may be assumed that ground water at this location contains levels of 2,6-DNT less than 8 µg/L. The estimated exposure level is:

$$\begin{aligned}\text{Exposure} &< \frac{8 \text{ } \mu\text{g/L} \times 2 \text{ L/day}}{70 \text{ kg}} \\ &< 0.23 \text{ } \mu\text{g/kg/day} \\ &< 2.3 \times 10^{-4} \text{ mg/kg/day}\end{aligned}$$

The carcinogenic risk associated with potential future consumption of this water is:



$$\begin{aligned}\text{Risk} &< 10^{-6} = \frac{2.3 \times 10^{-4} \text{ mg/kg/day}}{6.4 \times 10^{-7} \text{ mg/kg/day}} \\ &< 2.6 \times 10^{-4}\end{aligned}$$

Summary of Carcinogenic Risks--Assuming that carcinogenic risks associated with 2,4-DNT and 2,6-DNT are additive (ICF, Incorporated, 1985), the overall lifetime exposure carcinogenic risk to potential future residences using shallow ground water as a drinking water supply is less than 4.8×10^{-4} .

7.2.1.2 Noncarcinogenic Risks

Health risks for noncarcinogenic contaminants can be characterized by comparing expected exposure levels to acceptable intake levels (ADIs or AICs). The resulting hazard index provides a relative measure of the spread between acceptable and unacceptable exposure levels.

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exposure level exceeds the ADI (hazard index >1), the potential exists for adverse health effects for exposed populations. Estimation of exposure levels is performed according to the same methods and assumptions used for carcinogens.

The 2,4,6-TNT concentration in Well GW47 is 5.4 µg/L. The exposure level is:

$$\begin{aligned}\text{Exposure} &= \frac{5.4 \text{ µg/L} \times 2 \text{ L/day}}{70 \text{ kg}} \\ &= 0.15 \text{ µg/kg/day} \\ &= 1.5 \times 10^{-4} \text{ mg/kg/day}\end{aligned}$$

The hazard index is given by:

$$\begin{aligned}\text{HI} &= \frac{\text{estimated exposure}}{\text{acceptable daily intake}} \\ &= \frac{1.5 \times 10^{-4}}{1.4 \times 10^{-3}} \\ &= 0.11\end{aligned}$$

Because the hazard index is much less than 1, no adverse health effects would be expected. For this pathway, the hazard index is equivalent to the ratio between ground water concentration and the drinking water criteria presented in Table 6.3-1. Thus:

$$\begin{aligned}\text{HI} &= \frac{5.4 \text{ µg/L}}{50 \text{ µg/L}} \\ &= 0.11\end{aligned}$$

This simplified expression is used throughout this section for characterizing noncarcinogenic risks.

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The concentration of 1,3-DNB in GW47 is 2.3 µg/L. The acceptable drinking water criterion is 14 µg/L. Thus:

$$HI = \frac{2.3 \text{ µg/L}}{14 \text{ µg/L}}$$
$$= 0.16$$

Because the hazard index is much less than 1, no adverse health effects are expected as a result of exposure to 1,3-DNB in the Red Water Reservoirs area.

1,3,5-TNB was not detected at GW47 at a detection limit of 20 µg/L. This detection level is higher than normally achieved in other samples, resulting from dilution of the sample required to quantify the relatively high level of 2,4-DNT observed in this sample. 1,3,5-TNB was observed in other samples of ground water upgradient of GW47, and it seems likely that 1,3,5-TNB is present in ground water at this location. Thus:

$$HI < \frac{20 \text{ µg/L}}{200 \text{ µg/L}}$$
$$< 0.1$$

Because the hazard index is much less than 1, no adverse health effects associated with exposure to 1,3,5-TNB are expected in the Red Water Reservoirs area.

7.2.2 Environmental Impacts

The RI data indicate that the Red Water Reservoirs area does not result in exposure of aquatic biota or waterfowl, because contaminated sediments are covered by recent, clean sediments, resulting in no impacts on aquatic or terrestrial biota. Surface water concentrations of site-related contaminants are nondetectable, and the detection limits are less than criteria levels developed for protection of aquatic life.

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7.3 POND 13/WET WELL AREA

Based on the results of the RI and previous sections of this report, the No-Action Alternative is not viable for the Pond 13/Wet Well Area because of potential adverse effects on human health. Recreational bass fishermen and their families or friends with whom they share their catch might be exposed to unacceptable levels of DNTs if the pond were reopened for fishing, consistent with the management objectives of the State of West Virginia. The population at risk is small and difficult to quantify. Currently, Ponds 12 and 13 are closed to fishing. Pond 12 contains detectable levels of nitroaromatic contamination. Because the concentrations observed are below recommended criteria, Pond 12 could be reopened to fishing if it were routinely monitored for DNTs.

7.3.1 Human Health

7.3.1.1 Carcinogenic Risks

The RI identified two carcinogenic contaminants in surface waters of the Pond 13/Wet Well Area: 2,4-DNT and 2,6-DNT. Maximum contaminant levels were observed near the identified seep of contaminated ground water into Pond 13 at 5 µg/L of 2,6-DNT and 31 µg/L of 2,4-DNT. In other samples collected near the center of the pond and at the Pond 13 discharge to Pond 12, contaminant levels were lower: 2 µg/L of 2,4-DNT and <0.8 µg/L of 2,6-DNT (undetectable). Pond 12 contained 0.4 µg/L of 2,4-DNT, and 2,6-DNT was undetectable.

The risk associated with exposure to the DNTs in the Pond 13/Wet Well Area is based on the maximum contaminant levels observed in Pond 13 and assumes that Pond 13 was stocked and reopened to fishing.

Exposure factors for bass fishermen were presented in the EA for the first operable unit (ESE, 1986b) and are restated in Sec. 5.3. The exposure level is estimated by:

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$$\text{Exposure} = \frac{C_w \times W_f \times K_{wf}}{B_w}$$

where: C_w = concentration in pond water, maximum (31 $\mu\text{g/L}$);
 W_f = average mass of fish ingested per day by the
population at risk (kg/day);
 K_{wf} = water-to-fish partition coefficient (L/kg); and
 B_w = adult human body weight (70 kg).

K_{wf} = 6.67 L/kg (ESE, 1986b, Sec. 6.3.1.2)

W_f = 0.01 kg/day (refer to Sec. 5.3.3; also ESE, 1986b)

Thus:

$$\begin{aligned} \text{Exposure} &= \frac{31 \mu\text{g/L} \times 0.01 \text{ kg/day} \times 6.67 \text{ L/kg}}{70 \text{ kg}} \\ &= 0.030 \mu\text{g/kg/day} \\ &= 3.0 \times 10^{-5} \text{ mg/kg/day} \end{aligned}$$

The carcinogenic risk associated with bass consumption from Pond 13 is estimated by comparing the exposure level with the 2,4-DNT dose corresponding to a given level of risk (see Sec. 3.2.4):

$$\begin{aligned} \text{Risk} &< 10^{-6} \times \frac{3.0 \times 10^{-5} \text{ mg/kg/day}}{3.2 \times 10^{-6} \text{ mg/kg/day}} \\ &< 9.4 \times 10^{-6} \end{aligned}$$

The carcinogenic risk associated with consumption of bass potentially contaminated by 2,6-DNT from Pond 13 is calculated in a manner similar to that for 2,4-DNT. The 2,6-DNT exposure level is:

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$$\text{Exposure} = \frac{5 \mu\text{g/L} \times 0.01 \text{ kg/day} \times 6.67 \text{ L/kg}}{70 \text{ kg}}$$

$$= 4.8 \times 10^{-3} \mu\text{g/kg/day}$$

$$= 4.8 \times 10^{-6} \text{ mg/kg/day}$$

The risk is estimated as:

$$\text{Risk} < 10^{-6} \times \frac{4.8 \times 10^{-6} \text{ mg/kg/day}}{6.4 \times 10^{-7} \text{ mg/kg/day}} < 7.5 \times 10^{-6}$$

Summary of Carcinogenic Risks--Assuming that carcinogenic risks associated with 2,4-DNT and 2,6-DNT are additive (ICF, Incorporated, 1985), the overall lifetime exposure carcinogenic risk to bass fishermen using Pond 13 is $< 1.7 \times 10^{-5}$.

7.3.1.2 Noncarcinogenic Risks

Health risks for noncarcinogenic contaminants are determined by comparing observed contaminant levels with criteria designed for protection of human health using the hazard index as defined in Sec. 7.1.1.2. Water quality criteria designed for the protection of bass fishermen from exposure to noncarcinogenic nitroaromatics are presented in ESE (1986b, Sec. 6.3.1.2) and are: 2,4,6-TNT (1,700 $\mu\text{g/L}$); 1,3-DNB (1,000 $\mu\text{g/L}$); and 1,3,5-TNB (25,000 $\mu\text{g/L}$).

Observed concentrations of these contaminants in the Pond 13/Wet Well Area are highest in Pond 13 near the seep of contaminated ground water where 2,4,6-TNT is 32 $\mu\text{g/L}$ and 1,3-DNB is 0.4 $\mu\text{g/L}$. The hazard index is 0.02 or less for all noncarcinogenic compounds in the Pond 13/Wet Well Area, and no adverse health effects associated with these contaminants are anticipated.

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7.3.2 Environmental Impacts

The potential for adverse environmental impacts in the Pond 13/Wet Well Area is assessed by comparing observed maximum contaminant levels in Pond 13 with criteria designed to protect aquatic life. These criteria were derived from data analyzed by ESE (1986b, Sec. 6.3, Table 6.3-1).

The maximum observed concentration of 2,4,6-TNT is 32 µg/L, compared with the criterion of 60 µg/L based on freshwater chronic aquatic toxicity. The maximum concentration of 2,4-DNT observed was 31 µg/L, compared with the aquatic life chronic criterion of 230 µg/L. Observed concentrations of 1,3-DNB did not exceed 0.4 µg/L, compared with a criterion of 160 µg/L. No aquatic toxicity test data are available for 2,6-DNT; however, 2,6-DNT toxicity is assumed to be similar to that of 2,4-DNT (230 µg/L). The maximum observed concentration of 2,6-DNT (5 µg/L) is well below the chronic criterion for 2,4-DNT. 1,3,5-TNB was not detected in surface waters of the Pond 13/Wet Well Area, and the detection limit of 2 µg/L is below the chronic aquatic criterion of 80 µg/L.



In summary, the maximum observed concentrations of all nitroaromatic contaminants are below criteria developed for protection of freshwater aquatic biota for chronic exposures; therefore, no adverse environmental impacts are expected in the Pond 13/Wet Well Area. The ecological state of the site is observed on a regular basis by a trained wildlife biologist in the context of managing the McClintic Wildlife Station, and there is no evidence that aquatic or terrestrial biota have been harmed by the contamination.

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8.0 CONCLUSIONS

The presence of hazardous substances in each of the study areas evaluated in this EA report presents an endangerment to public health or the environment. Endangerment in each of the areas is associated with potential future land or water uses. No existing human or animal population is currently exposed to unacceptable levels of contamination. In the Pond 13/Wet Well Area, unacceptable exposure is prevented by prohibiting fishing in Pond 13. This action was taken in response to the discovery of contamination; fishing would be encouraged in this area if the water were uncontaminated. Although no-fishing signs are posted, access to Pond 13 is unrestricted.

With the exception of the Pond 13/Wet Well Area, the fact that there is negligible exposure to contamination results from the historical land and water uses in the area. If these traditional patterns of resource utilization were simply extrapolated, it could be concluded that unacceptable exposures are unlikely in the future. For example, water supply wells in the area traditionally are completed in the deep aquifer, which is not expected to become contaminated in excess of safe drinking water levels. A municipal water supply, which is not threatened by site contamination, is available to existing and new residences. However, there are no enforceable prohibitions or feasibility constraints that would prevent use of the surficial aquifer, which is unacceptably contaminated in both the Red Water Reservoirs and the Acids Area/Yellow Water Reservoir.

Concern regarding ground water contamination is more appropriate for the Red Water Reservoirs than for the other study areas. Although the precise extent of contaminated ground water in this area is unknown, it is likely that approximately 30 acres west of the McClintic Wildlife Station and east of SR 62 exhibits unacceptable ground water contamination. This area, currently in mixed agricultural and residential use, has experienced increasing residential development in

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the last 10 years. Municipal water is available to this area. Ground water contamination in the Pond 13/Wet Well Area is confined to the McClintic Wildlife Station where substantial institutional constraints prevent future development of the ground water resource. Residential development and potable use of ground water in the Acids Area/Yellow Water Reservoir are unlikely, considering the planned industrial/commercial development of this area which is consistent with existing use of adjacent parcels.

Surface and subsurface soils in the Acids Area/Yellow Water Reservoir exhibit contaminant levels that could result in unacceptable exposure to workers if the soils were disturbed and/or future industrial use resulted in dust generation.

Environmental effects resulting from observed contamination levels are apparently minimal. Vegetation stress is observed in a small (100 ft²) portion of the Acids Area/Yellow Water Reservoir. Surficial soils on McClintic Wildlife Station (see ESE, 1986b) are contaminated to higher levels over a larger area, with no observable effect on wildlife. Surface water criteria designed to protect aquatic biota are not exceeded in Pond 13. Contaminated sediment in the Red Water Reservoirs is isolated by more recent, clean sediments with no pathway for exposure of aquatic life, water fowl, or terrestrial biota. There is no evidence indicating that there are significant impacts on threatened and endangered species.

In all areas, the critical contaminants are 2,4-DNT and 2,6-DNT. The presence of noncarcinogen contaminants (e.g., 2,4,6-TNT) poses a significant endangerment only in the Acids Area/Yellow Water Reservoir. Criteria for soils are presented in terms of total nitroaromatic contamination, because a simple field analysis technique for total nitroaromatics appears to be a useful analytical procedure for determining the precise locations of actionable soils and remediation.

The development of these criteria was based on the assumption that characteristic ratios exist between specific compounds and total nitroaromatics. The total nitroaromatic cleanup criteria recommended in this document were developed to protect populations at risk from unacceptable exposure to each of the nitroaromatic contaminants that comprise the total. Development of criteria for soils and sediments based on this simple and inexpensive field analytical procedure provides guidance useful for the calculation of volumes of soils and sediments exceeding these criteria since more data are available for total nitroaromatic contamination from the RI survey. This analytical procedure may also be used efficiently in the field during implementation of remedial actions selected after completion of the feasibility study.

Some of the findings of this EA, as well as the EA for the first operable unit (ESE, 1986b), are uncertain. The greatest source of uncertainty is derived from the the lack of valid data on the concentration of site contaminants in tissue of game animals and fish. These data gaps are particularly significant at this site because recreational hunting and fishing, and consumption of the harvest, are important exposure pathways at McClintic Wildlife Station. Tissue analyses were not performed because of recognized difficulties in QA/QC and because of potential difficulties in data interpretation. To complete the exposure assessment involving these pathways, conservative assumptions were adopted, particularly assumptions regarding equilibrium partitioning and no metabolic transformation. Thus, it is believed that the risks are not underestimated, and the recommended remedial objectives are sufficiently protective of human health and the environment.

It is recommended that tissue samples be collected and analyzed for nitroaromatic contaminants as part of the remedial design phase if valid sampling and analysis protocols can be developed and certified with respect to recognized QA/QC specifications. It is unlikely that the valid tissue analyses will show that exposure via hunting and fishing has

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been underestimated. Rather, it can be anticipated that valid tissue analyses may reveal that exposure via hunting and fishing has been overestimated and that the remedial objectives are unnecessarily stringent.

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APPENDIX A

DERIVATION OF LATERAL DISPERSION EQUATION

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DERIVATION OF LATERAL DISPERSION EQUATION

Lateral hydrodynamic dispersion is treated as a Fickian dispersion process (Csanady, 1973), which is expressed,

$$\sigma = \sqrt{2 D t}$$

where: σ = standard deviation of the dispersing
contaminant (ft),

D = dispersion coefficient (ft²/day), and

t = time of travel (days).

Hydrodynamic dispersion through porous media (Freeze and Cherry, 1979) is expressed:

$$D = \alpha V_1$$

where: α = hydrodynamic dispersivity (ft), and

V_1 = average linear velocity of ground water (ft/day).

The time of travel is simply:

$$t = \frac{X}{V_1}$$

where: X = distance from source to receptor (ft).

According to Gelhar et al. (1985):

$$\alpha \approx 0.033 X$$

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Substituting into the equation for σ :

$$\begin{aligned}\sigma &= \sqrt{\frac{2 (0.033 X) V_1 X}{V_1}} \\ &= \sqrt{0.066 X^2} \\ &= 0.26 X\end{aligned}$$

In Fickian dispersion, more than 99 percent of the contaminant will be contained within $\pm 2 \sigma$ of the plume centerline. Therefore, the full width of the plume is $4 \sigma = 1.03 X$. This dispersed width must be added to the initial width of the source area to obtain the full width of the plume. Thus:

$$W_r = W_s + 1.03 X$$

where: W_s = width of the source area, and

W_r = width at the receptor location.

For longitudinal dispersion, the relationship is derived in the same way. The longitudinal dispersion coefficient, σ_1 , however, is given by (Gelhar et al., 1985):

$$\begin{aligned}\sigma_1 &= 0.1 X \\ \text{Thus: } \sigma_1 &= \sqrt{\frac{\sigma (0.1 X) V_1 X}{V_1}} \\ &= \sqrt{0.2 X^2} \\ &= 0.45 X\end{aligned}$$

The plume length at the receptor, L_r , is given by:

$$L_r = L_s + 1.79 X$$

For vertical dispersion, similar relationships may be derived using:

$$\sigma_v = 2.5 \times 10^{-4} X$$

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